PERFORMANCE REPORT FOR FEDERAL AID GRANT F-63-R, SEGMENT 4

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MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT INVESTIGATIONS

Maryland Department of Natural Resources

Fisheries Service

Tawes State Office Building B-2

Annapolis, Maryland 21401

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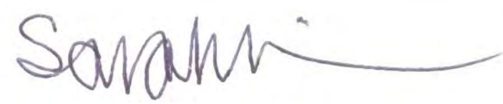
Approval



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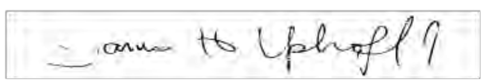

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Report Organization

This report consists of summaries of activities for Jobs 1–4 under this grant. All pages are numbered sequentially; there are no separate page numbering systems for each Job. Job activities are reported in separate numbered sections. For example, Job 1, Section 1 would cover development reference points (Job 1) for stream spawning habitat of anadromous fish (Section 1). Tables in a Job are numbered as section number – table number (1-1, 1-2, etc). Figures are numbered in the same fashion. This nomenclature applies to Job 1.

Throughout the report, multiple references to past annual report analyses are referred to. The complete PDF versions of many past annual reports can be found under the Publications and Report link on the Fisheries Habitat and Ecosystem (FHEP) website page on the Maryland DNR website. The website address is http://dnr.maryland.gov/fisheries/Pages/FHEP/index.aspx.

Table of Abbreviations for Jobs 1-3

°C	Celsius, temperature
α	Level of significance
μ (micron)	micrometer or one millionth of a meter
μg/L	Micrograms per liter
μmho/cm or μS/cm	Conductivity measurement as micromhos per centimeter or micro-Siemens per centimeter.
A	Area
A/ha	Structure area per hectare
AM	Arithmetic mean
ASMFC	Atlantic States Marine Fisheries Commission
BI	Blue Infrastructure
BRP	Biological reference point
C	Structures in a watershed
C / ha	Structure counts per hectare
CAD	Computer Aided Design
CBP	Chesapeake Bay Program
cfs	Cubic feet per second, measurement of flow volume
CI	Confidence Interval
COL	Cooperative Oxford Laboratory, NOAA
CPE	Catch per effort
CV	Flow coefficient of variation
DO	Dissolved oxygen
EBFM	Ecosystem-Based Fisheries Management
ER	Environmental Review Program in MD DNR
ESRI	Environmental Systems Research Institute
FERC	Federally Energy Regulatory Commission
FIBI	Fish Index of Biological Integrity (see reference Morgan et al. 2007)
GIS	Geographic Information System
gm	Gram
GM	Geometric mean
ha	Hectares
hr	Hour
P_i	Proportion of samples with target species *i*
IA	Impervious surface area estimated in the watershed
in	Inches
IS	Impervious surface
ISRPs	Impervious surface reference points
km	Kilometer
km^2	Square kilometers

L_P	Proportion of Tows with yellow perch larvae during a standard time period and where larvae would be expected
M	Median flow
m	Meter
Max	Maximum
MD DNR	Maryland Department of Natural Resources
MDE	Maryland Department of Environment
MDP	Maryland Department of Planning
mg/L	Milligrams per liter
Min	Minimum
mm	Millimeter
MT	Metric ton
$N_{present}$	Number of samples with herring eggs and-or larvae present
N_{total}	Total sample size
N	Sample size
NAD	North American Datum
NAJFM	North American Journal of Fisheries Management
N_i	Number of samples containing target species
NOAA	National Oceanic and Atmospheric Administration
NRC	National Research Council
OM	Organic matter
OM0	Proportion of samples without organic matter
P or α	Level of significance
P_{herr}	Proportion of samples where herring eggs and-or larvae were present
Pclad	Proportion of guts with cladocerans
Pcope	Proportion of guts with copepods
Pothr	Proportion of guts with “other” food items
P_0	Proportion of guts without food
P_i	Proportion of samples with a target species
pH	Concentration of hydrogen ions; the negative base-10 logarithm of hydrogen ion concentration.
ppt or ‰	Parts per thousand, salinity measurement unit
P_{Qwp}	Proportion of samples with White Perch > 200 mm TL
P_{75th}	Proportion in the upper quartile
P_{25th}	Proportion in the lower quartile
QA	Quality assurance
r	Correlation coefficient, statistical measurement
RKM	River kilometer
SAS	Statistical Analysis System
SAV	Submerged aquatic vegetation
SD	Standard deviation
SE	Standard error

TA	Estimate of total area of the watershed
TAN	Total ammonia nitrogen
TEA	Tidal Ecosystem Assessment Division in MD DNR
TL	Total length
USACOE	United States Army Corps of Engineers
USFWS	United States Fish and Wildlife Service
USGS	United States of Geological Service
$\mathbf{V}_{\mathbf{target}}$	Percentage of DO measurements that met or fell below the 5 mg/L target
$\mathbf{V}_{\mathbf{threshold}}$	Percentage of DO measurements that fell at or below the 3 mg/L threshold

Definitions

Alosines	American Shad, Hickory Shad, Blueback Herring, and Alewife are Alosines, which belong to the Herring family, Clupeidae.
Anadromous Fish (Spawning)	Fish, such as Shad, Herring, White Perch, and Yellow Perch, ascend rivers from the Chesapeake Bay or ocean for spawning.
Brackish Water	Water that has more salinity than freshwater. The salinity of brackish water is between 0.5 – 30 ppt.
Coastal Plain	An area underlain by a wedge of unconsolidated sediments including gravel, sand, silt and clay and is located in the eastern part of Maryland, which includes the Chesapeake Bay's eastern and western shores, up to the fall line roughly represented by U.S. Route 1.
Development	Refers to land used for buildings and roads.
Estuary	A body of water in between freshwater and the ocean; an estuary can be subject to both river and ocean influences, such as freshwater, tides, waves, sediment, and saline water.
Finfish	Referring to two or more species of fish and excludes shellfish.
Floodplain	Refers to land that is adjacent to a stream or river that experiences flooding during periods of high flow.
Fluvial	Of or pertaining to rivers.
Hypoxia	Occurrence of low oxygen conditions.
Icthyoplankton	Refers to the eggs and larvae of fish.
Impervious surface (IS)	Hard surfaces that are not penetrated by water such as pavement, rooftops, and compacted soils.
Mesohaline	A region within an estuary with a salinity range between 5 and 18 ppt.
Non-Tidal Waters (Stream)	Areas that are not influenced by tides.
Oligohaline Subestuary	A brackish region of an estuary with a salinity range between 0.5 and 5 ppt.
Piedmont	A plateau region located in the eastern United States and is made up of low, rolling hills that contain clay-like and moderately fertile soils.
Planktivores	Animals that feed primarily on plankton (organisms that float within the water column).
Richness	The number of different species represented in a collection of individuals.
Riparian zone	The area between land and a river and/or stream, also known as a river bank.

Rural	Referring to areas undeveloped such as farmland, forests, wetlands and areas with low densities of buildings.
Stock Assessments	Assessments of fish populations (stocks); studies of population dynamics (abundance, growth, recruitment, mortality, and fishing morality).
Stock Level	Refers to the number or population weight (biomass) of fish within a population.
Subestuary	A smaller system within a larger estuary such as a branching creek or tributary within the estuary.
Suburb	An area that has mostly residential development located outside of city or town boundaries.
Threshold	A breaking point of an ecosystem and when pressures become extreme can produce abrupt ecological changes.
Tidal-Fresh Subestuary	An area containing mainly freshwater with salinity less than 0.5 ppt, but tidal pulses may bring higher salinity.
Tidal Waters	Waters influenced by tides.
Urban	A developed area characterized by high population, building, and road densities; may be considered a city or town.
Urbanization	Process of conversion of rural land to developed land.
Watershed	Defines a region where all of the water on and under the land drains into the same body of water.
Wetlands	An area of ground that is saturated with water either permanently or seasonally; they have unique vegetation and soil conditions and can either be saltwater, freshwater, or brackish depending on location.
Zooplankton	Animals that drift within the water column; these animals are typically very small, but may be large (jellyfish and comb jellies).

Job 1: Development of habitat-based reference points for recreationally important Chesapeake Bay fishes of special concern: development targets and thresholds

Jim Uphoff, Margaret McGinty, Alexis Maple, Carrie Hoover and Paul Parzynski

Executive Summary

Stream Ichthyoplankton - Proportion of samples with Herring (Blueback Herring, Alewife, American Shad, and Hickory Shad) eggs and-or larvae (P_{herr}) provided a reasonably precise estimate of relative abundance based on encounter rate. Magnitude of P_{herr} may indicate how much habitat is available or how attractive it is from year to year more-so than abundance of spawners. In developed watersheds, a combination of urban and natural stream processes may create varying amounts of ephemeral spawning habitat annually and dampen spawning migrations through increased conductivity (primarily from road salt). Regression analyses indicated significant and logical relationships among $P_{herr,}$ level of development (structures per hectare or C / ha), and conductivity (a measure of dissolved salts in water) consistent with the hypothesis that urbanization was detrimental to stream spawning. An unavoidable assumption of these analyses was that watersheds at different levels of development could substitute for time-series.

In Mattawoman Creek, we obtained adequate sample sizes by pooling data across years to estimate proportions of samples with White Perch eggs and larvae or Yellow Perch larvae. This allowed us to compare for 1989-1991 collections (C / ha = 0.43–0.47) with 2008-2013 (C / ha = 0.87-0.91) at the same combinations of downstream sites. These estimates did not detect a loss in stream spawning for Yellow Perch. A decline in White Perch stream spawning was likely.

Estuarine Yellow Perch Larval Sampling - Estimates of the proportion of plankton net tows with Yellow Perch larvae, L_p, declined perceptibly once watershed development exceeded the threshold (0.83 structures per hectare, C / ha, equivalent to 10% impervious surface, IS). A forest cover classification in a watershed was associated with higher L_p (median L_p= 0.79) than agriculture (median L_p= 0.51) or development (median L_p= 0.30). Interpretation of the influence of salinity class or primary land cover on L_p needs to consider that our survey design was limited to existing patterns of development. All estimates of L_p at or below target levels of development (0.27 C / ha or 5% IS; rural forested and agricultural watersheds) or at and beyond high levels of development (1.59 C / ha or 15% IS; suburban and urban watersheds) were from brackish subestuaries; estimates of L_p for development between these levels were from tidal-fresh subestuaries with forested watersheds.

There appears to be some potential for development to influence organic matter (OM) and larval Yellow Perch feeding dynamics. However, OM may not matter much if there is not a match in the timing of copepod abundance and early feeding stages of Yellow Perch larvae. We did not interpret RNA/DNA ratios as rejecting or supporting the OM hypothesis since there was little indication of a match of zooplankton and Yellow Perch larvae in 2012 (primarily upper Bay subestuaries) or 2013 (primarily Potomac River subestuaries). A contrasting year of high overall feeding success would greatly aid interpretation of RNA/DNA ratios. Our RNA/DNA sampling indicated that most Yellow Perch larvae collected were in the starved category in both years (55 of 91 larvae in 2012 and 2013 (137 of 170).

Estuarine Fish Community Sampling - Plots of species richness (number of species encountered) against our indicator of watershed development (structures per hectare or C / ha) in 4.9 m trawl collections did not suggest relationships for either tidal-fresh or oligohaline (low salinity) subestuaries. Plots did suggest that number of species declined when development went beyond the threshold in watersheds of mesohaline (mid-strength salinity) subestuaries. In general these exploratory analyses of species richness and development supported trends found in analyses of development and DO. Bottom DO was not negatively influenced by development in tidal-fresh or oligohaline subestuaries, but was in mesohaline subestuaries. Depletion of DO in bottom waters of mesohaline subestuaries to hypoxic or anoxic levels represented a direct loss of habitat. Availability of White Perch at a size of interest to anglers ($\geq$ 200 mm TL) were more likely to be high in mesohaline subestuaries with rural or transition watersheds, and least likely to be found in subestuaries with suburban-urban watersheds independent of salinity class.

We continued to track bottom dissolved oxygen (DO), submerged aquatic vegetation (SAV), finfish abundance and number of finfish species collected in 3.1 m and 4.9 m trawl samples from Mattawoman Creek and compared them to changes in C / ha. For this report, we obtained measurements of total ammonia nitrogen (TAN; NH_3 plus NH_4) from a Chesapeake Bay Program (CBP) monitoring site located in the channel adjacent to a continuous monitor within dense SAV bed.

The level of development in Mattawoman Creek's watershed more than doubled between 1989 (0.43 C / ha) and 2011 (0.91 C / ha) and reached the suburban threshold in 2006. A downward shift of bottom DO after 2000 corresponded to changes in Mattawoman

Creek's subestuary chlorophyll a from high to low and shift in SAV acreage from low (coverage of ~10% or less of water area) to high (coverage of > 30%). Median TAN was low and stable through 2000 and then began a rapid rise to a spike in 2002. Median TAN dropped after 2002, but was elevated beyond that seen prior to 2001; during 2007-2009, median TAN was consistently elevated beyond this period's baseline.

We developed a hypothesis that water quality dynamics in Mattawoman Creek's extensive SAV beds (low DO, high pH, and high organic matter) may be creating episodes of ammonia toxicity for fish. Mattawoman Creek's finfish abundance appeared to be susceptible to boom and bust dynamics after 2001. "Busts" were concurrent with spikes (2002) or plateaus (2007-2009) of TAN. Collapses of the magnitude exhibited during 2002 and 2008-2009 were not detected previously.

Job 1: Introduction

Fisheries management uses biological reference points to determine how many fish can be safely harvested from a stock (Sissenwine and Shepherd 1987). The primary objective of Project 1 was to devise reference points for development as a similar tool for fish habitat management. Creating reference points that indicate safe and unsafe watershed stress from development involves determining functional relationships between an indicator of watershed development and habitat quality (water quality, physical structure, etc) or a species response (habitat occupation, abundance, distribution, mortality, recruitment success, growth, fish condition, etc). Quantitative, habitat-based reference points are envisioned as a basis for strategies for managing fisheries in increasingly urbanizing coastal watersheds and for communicating the limits of fisheries resources to withstand development-related habitat changes to stakeholders and agencies involved in land-use planning.

Maryland Fisheries Service has chosen counts of structures per watershed hectare from Maryland Department of Planning property tax map data as our indicator of development. Tax map indicators are standardized, annually updated, readily accessible and based on observed quantities (structure counts or structure area), and are strongly related to impervious surface estimated from satellite images (Uphoff et al. 2012).

Fisheries managers in Maryland do not have authority to manage land-use, so they need to consider managing fish differently at different levels of development if productivity diminishes. The target level of development for fisheries is indicated by about 0.27 structures per hectare (C / ha) or less (~ 5% impervious surface or IS; Uphoff et al. 2012). This target level of development in Maryland is characterized by forests, working farms, and wetlands that support productive fish habitat and fisheries. Land-use at this level does not undermine effectiveness of harvest controls for sustaining fish populations. Conserving watersheds at this level of development would be ideal. Once above this level of development, increasing consideration has to be given to habitat conservation, watershed revitalization (small scale ecological re-engineering), and watershed reconstruction (large scale ecological re-engineering). Revitalization and reconstruction could consist of measures such as road salt management, stemming leaks in sewage pipes, improving septic systems, stormwater retrofits, stream rehabilitation, replenishment of riparian buffers, creation of wetlands, planting upland forests, and "daylighting" of buried streams. Lowering harvest levels may be able to offset habitat degradation, but places the burden of development on anglers.

The threshold of development of 0.83 C / ha (10% IS) represents a suburban landscape where serious aquatic habitat degradation becomes apparent (Uphoff et al. 2012). At this point, conservation of remaining natural lands and habitat revitalization and reconstruction will be the primary tools for fishery sustainability. Harvest restrictions may be ineffective in stemming fishery declines. By 1.59 C / ha (15% IS), serious habitat problems make fish habitat revitalization very difficult. Managers must deal with substantially less productive fisheries.

Job 1 activities in 2013 included spring stream anadromous fish icthyoplankton collections, spring yellow perch larval presence-absence sampling, and summer sampling of estuarine fish communities and habitat. These activities are reported as separate sections in Job 1. These efforts were collectively aimed at defining the impact of development on target fish species populations and habitats and judging how development reference points proposed by Uphoff et al. (2011a) for brackish subestuaries (based on dissolved oxygen and habitat occupation by juveniles and adults of our target species) apply to Tax Map data, and other life stages and habitats.

References:

Uphoff, J.H., Jr., M. McGinty, R. Lukacovic, J. Mowrer and B. Pyle. 2011a. Impervious surfaces, summer dissolved oxygen and fish distribution in Chesapeake Bay subestuaries: linking watershed development, habitat and fisheries management. North American Journal of Fisheries Management 31(3):554-566.

Uphoff, J.H., Jr., and coauthors. 2012. Marine and estuarine finfish ecological and habitat investigations. Performance Report for Federal Aide Grant F-63-R, Segment 2. Maryland Department of Natural Resources, Annapolis, Maryland.

Job 1, Section 1: Stream Ichthyoplankton Sampling

Introduction

Surveys to identify spawning habitat of White Perch, Yellow Perch and "Herring" (Blueback Herring, Alewife, American Shad, and Hickory Shad) were conducted in Maryland during 1970-1986. These data were used to develop statewide maps depicting anadromous fish spawning habitat (O'Dell et al. 1970; 1975; 1980; Mowrer and McGinty 2002). Many of these watersheds have undergone considerable development and recreating these surveys provided an opportunity to explore whether spawning habitat declined in response to urbanization. Surveys based on the sites and methods of O'Dell et al. (1975) were used to sample Mattawoman Creek (2008-2013), Piscataway Creek (2008-2009 and 2012-2013), Bush River (2005-2008) and Deer Creek (2012-2013; Figure 1-1).

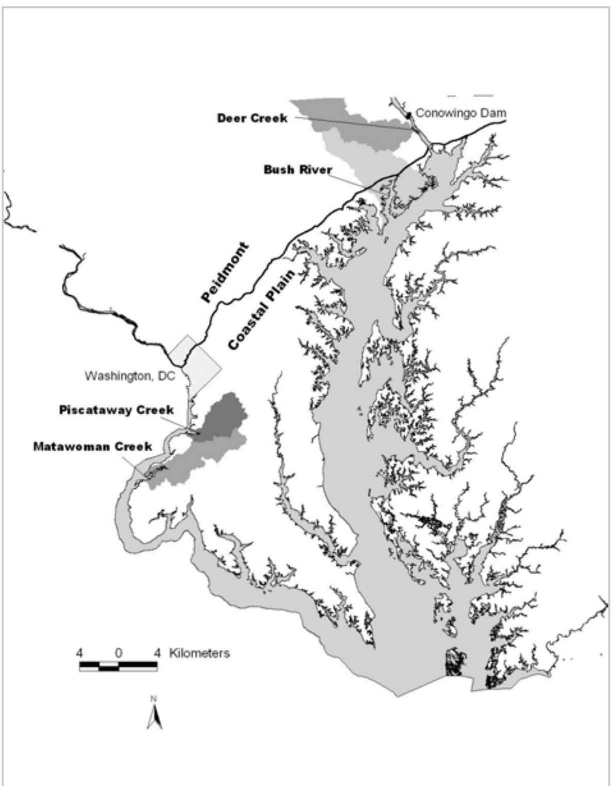

Figure 1-1. Watersheds sampled for stream spawning anadromous fish eggs and larvae in 2005-2013. Coastal Plain and Piedmont Regions are indicated.

Mattawoman and Piscataway Creeks are adjacent Coastal Plain watersheds along an urban gradient emanating from Washington, DC (Figure 1-1). Piscataway Creek's watershed is both smaller than Mattawoman Creek's and closer to Washington, DC. Bush River is located in the urban gradient originating from Baltimore, Maryland, and is located in both the Coastal Plain and Piedmont physiographic provinces. Deer Creek is entirely located in the Piedmont north of Baltimore, near the Pennsylvania border (Figure 1-1; Clearwater et al. 2000).

We developed two indicators of anadromous fish spawning in a watershed based on presence-absence of eggs and larvae: occurrence at a site and proportion of samples with eggs and larvae. Occurrence of eggs or larvae of an anadromous fish group (White Perch, Yellow Perch, and Herring) at a site, recreated the indicator developed by O'Dell et al. (1975; 1980). This indicator was compared to the extent of development in the watershed (counts of structures per hectare or C/ha) between the 1970s and the present. We also developed an indicator of relative abundance, proportion of samples with eggs and-or larvae of anadromous fish groups, from collections in the 2000s and compared it to C/ha and summarized conductivity data. Conductivity was monitored and examined to see whether urbanization had affected stream water quality. Increases in conductivity have been strongly associated with urbanization (Wang and Yin 1997; Paul and Meyer 2001; Wenner et al. 2003; Morgan et al. 2007; Carlisle et al. 2010; Morgan et al. 2012).

Methods

Stream sites sampled for the anadromous fish eggs and larvae during 2005-2013 were typically at road crossings that O'Dell et al. (1975) determined were anadromous fish spawning sites during the 1970s. O'Dell et al. (1975) summarized spawning activity as the presence of any species group egg, larva, or adult at a site. Eggs and larvae were sampled from stream drift ichthyoplankton nets and adults were sampled by wire traps. All collections during 2005-2013, with the exception of Deer Creek during 2012-2013, were made by citizen volunteers who were trained and monitored by program biologists. During March to May, between 2008 and 2013, ichthyoplankton samples were collected in Mattawoman Creek from three tributary sites (MUT3-MUT5) and four mainstem sites (MC1-MC4; Figure 1-2; Table 1-1). Tributary site (MUT4) was selected based on volunteer interest and added in 2010. Piscataway Creek stations were sampled during 2008-2009 and 2012-2013 (Figure 1-3; Uphoff et al. 2010). Bush River stations were sampled during 2005-2008 (Figure 1-4; McGinty et al. 2009). Deer Creek sites SU01-SU04 were added to sampling in 2012 and sampling continued in 2013 with the addition of site SU05 Figure 1-5). Table 1-1 summarizes sites, dates, and sample sizes in Mattawoman, Piscata-

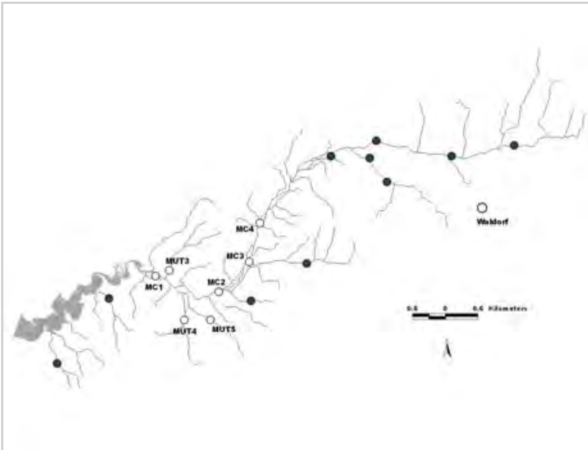

Figure 1-2. Mattawoman Creek's 1971 and 2008-2013 sampling stations.

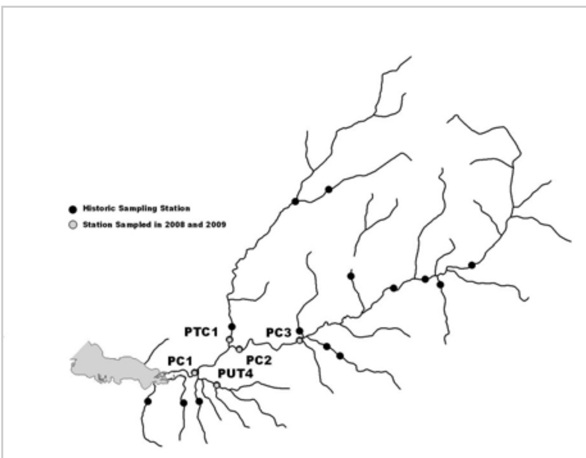

Figure 1-3. Piscataway Creek's 1971, 2008-2009, and 2012-2013 sampling stations.

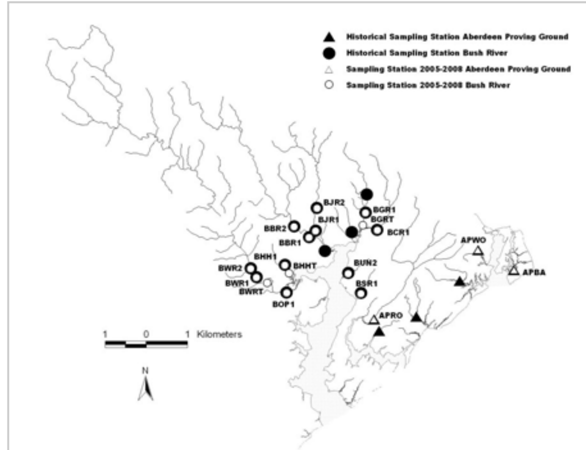

Figure 1-4. Bush River 1973 and 2005-2008 sampling Stations on Aberdeen Proving Ground were not included with Bush River data.

Figure 1-5. Deer Creek's 1972 and 2012-2013 sampling stations.

way and Deer Creeks, and Bush River during 2005-2013.

Ichthyoplankton samples were collected at each site using stream drift nets constructed of 360-micron mesh. Nets were attached to a square frame with a 300 460 mm opening. The stream drift net configuration and techniques were the same as those used by O'Dell et al. (1975). The frame was connected to a handle so that the net could be held stationary in the stream. A threaded collar on the end of the net connected a mason jar to the net. Nets were placed in the stream for five minutes with the opening facing upstream. Nets were retrieved and rinsed in the stream by repeatedly dipping the lower part of the net and splashing water through the outside of the net to avoid sample contamination. The jar was removed from the net and an identification label describing site, date, time, and collectors was placed in the jar. The jar was sealed and placed in a cooler with ice for transport when collections were made by volunteers. Preservative was not added by volunteers at a site because of safety and liability concerns. Formalin was added on site by DNR personnel. Water temperature (°C), conductivity (µS/cm), and dissolved oxygen (DO, mg/L) were recorded at each site using a hand-held YSI Model 85 meter. Meters were calibrated for DO each day prior to use. All data were recorded on standard field data forms and verified at the site by a volunteer. After a team finished sampling for the day, the samples were preserved with 10% buffered formalin. Approximately 2-ml of rose bengal dye was added in order to stain the organisms red to aid sorting.

Ichthyoplankton samples were sorted in the laboratory by project personnel. All samples were rinsed with water to remove formalin and placed into a white

Subestuary	Year	Number of Sites	1st Sampling Date	Last Sampling Date	Number of Dates	N
Bush	2005	13	18-Mar	15-May	16	99
Bush	2006	13	18-Mar	15-May	20	114
Bush	2007	14	21-Mar	13-May	17	83
Bush	2008	12	22-Mar	26-Apr	17	77
Piscataway	2008	5	17-Mar	May	8	39
Piscataway	2009	6	9-Mar	1May	11	60
Piscataway	2012	5	5-Mar	16-May	11	55
Piscataway	2013	5	11-Mar	28-May	11	55
Mattawoman	2008	9	8-Mar	9-May	10	90
Mattawoman	2009	9	8-Mar	11-May	10	70
Mattawoman	2010	7	7-Mar	15-May	11	75
Mattawoman	2011	7	5-Mar	15-May	14	73
Mattawoman	2012	7	Mar	13-May	11	75
Mattawoman	2013	7	10-Mar	25-May	12	80
Deer	2012	4	20-Mar	7-May	11	44
Deer	2013	5	19-Mar	23-May	19	87

Table 1-1. Summary of subestuaries, years sampled, number of sites, first and last dates of sampling, and stream ichthyoplankton sample sizes (N).

sorting pan. Samples were sorted systematically (from one end of the pan to another) under a 10x bench magnifier. All eggs and-or larvae were removed and were retained in a small vial with a label (site, date, and time) and stored with 20% ethanol for later identification under a microscope. Each sample was sorted systematically a second time for quality assurance (QA). Any additional eggs and-or larvae found were removed and placed in a vial with a label (site, date, time, and QA) and stored with 20% ethanol for identification under a microscope. All eggs and larvae found during sorting (both in original and QA vials) were identified as either Herring (Blueback Herring, Alewife, Hickory Shad, and American Shad), Yellow Perch, White Perch, unknown (eggs and-or larvae that were too damaged to identify) or other (indicating another fish species) and a total count (combining both original and QA vials) for each site was recorded, as well as the presence and absence of each of the above species. The four Herring species' eggs and larvae are very similar (Lippson and Moran 1974) and identification to species can be problematic.

We used property tax map based counts of structures in a watershed, standardized to hectares (C/ha), as our indicator of development (Uphoff et al. 2012). This indicator has been provided to us by Marek Topolski of the Fishery Management Planning and Fish Passage Program. Tax maps are graphic representations of individual property boundaries and existing structures that help State tax assessors locate properties (Maryland Department of Planning or MDP 2013). All tax data were organized by county. Since watersheds straddle political boundaries, one statewide tax map was created for each year of available tax data, and then subdivided into watersheds. Maryland's tax maps are updated and maintained electronically as part of MDP's Geographic Information System's (GIS) database. Files were managed and geoprocessed in ArcGIS 9.3.1 from Environmental Systems Research Institute (ESRI 2009). All feature datasets, feature classes, and shapefiles were spatially referenced using the NAD_1983_StatePlane_Maryland_FIPS_1900 projection to ensure accurate feature overlays and data extraction. ArcGIS geoprocessing models were developed using Model Builder to automate assembly of statewide tax maps, query tax map data, and assemble summary data. To create watershed land tax maps, each year's statewide tax map was clipped using the

MD 8-digit watershed boundary file; estuarine waters were excluded. These watershed tax maps were queried for all parcels having a structure built from 1700 to the tax data year. A large portion of parcels did not have any record of year built for structures, but consistent undercounts should not have presented a problem since we were interested in the trend and not absolute magnitude (Uphoff et al. 2012).

Uphoff et al. (2012) developed an equation to convert annual estimates of C/ha to estimates of impervious surface (IS) calculated by Towson University from 1999-2000 satellite imagery. Estimates of C/ha that were equivalent to 5% IS (target level of development for fisheries; a rural watershed), 10% IS (development threshold for a suburban watershed), and 15% IS (highly developed suburban watershed) were estimated as 0.27, 0.83, and 1.59 C/ha, respectively (Uphoff et al. 2012).

Estimates of C/ha were available from 1950 through 2011 (M. Topolski, MD DNR, personal communication). Estimates of C/ha for 2011 were used to represent 2012 and 2013.

Mattawoman Creek's watershed equaled 25,168 ha and estimated C/ha was 0.85-0.91 during 2008-2013; Piscataway Creek's watersheds equaled 17,999 ha and estimated C/ha was 1.37-1.46 during 2008-2013; and Bush River's watershed equaled 39,644 ha and estimated C/ha was 1.37-1.45 during 2005-2008; (M. Topolski, MD DNR, personal communication). Deer Creek (Figure 1-1), a tributary of the Susquehanna River, was added in 2012 as a spawning stream with low watershed development (watershed area = 37,701 ha and development level = 0.24 C/ha; (M. Topolski, MD DNR, personal communication). It was sampled in 2012-2013 by DNR biologists from the Fishery Management Planning and Fish Passage Program at no charge to this grant.

Conductivity measurements collected for each date and stream site (mainstem and tributaries) during 2008 -2013 from Mattawoman Creek were plotted and mainstem measurements were summarized for each year. Unnamed tributaries were excluded from calculation of summary statistics to capture conditions in the largest portion of habitat. Comparisons were made with conductivity minimum and maximum reported for Mattawoman Creek during 1991 by Hall et al. (1992). Conductivity data were similarly summarized for Piscataway Creek mainstem stations during 2008-2009 and 2012-2013. A subset of Bush River stations that were sampled each year during 2005-2008 (i.e., stations in common) were summarized; stations within largely undeveloped Aberdeen Proving Grounds were excluded because they were not sampled every year. Conductivity was measured with each sample in Deer Creek in 2012-2013.

A water quality database maintained by DNR's Tidewater Ecosystem Assessment (TEA) Division (S. Garrison, MD DNR TEA, personal communication) provided conductivity measurements for Mattawoman Creek during 1970-1989. These historical measurements were compared with those collected in 2008-2013 to examine changes in conductivity over time. Monitoring was irregular for many of the historical stations. Table 1-2 summarizes site location, month sampled, total measurements at a site, and what years were sampled. Historical stations and those sampled in 2008-2013 were assigned river kilometers (RKM) using a GIS ruler tool that measured a transect approximating the center of the creek from the mouth of the subestuary to each station location. Stations were categorized as tidal or non-tidal. Conductivity measurements from eight non-tidal sites sampled during 1970-1989 were summarized as monthly medians. These sites bounded Mattawoman Creek from its junction with the estuary to the city of Waldorf (Route 301 crossing), the major urban influence on the watershed.

Historical monthly median conductivities at each mainstem Mattawoman Creek non-tidal site were plotted with 2008-2013 spawning season median conductivities.

Presence of White Perch, Yellow Perch, and Herring eggs and-or larvae at each station in 2013 was compared to past surveys to determine which sites still supported spawning. We used the criterion of detection of eggs and-or larvae at a site (O'Dell et al. 1975) as evidence of spawning. Raw data from early 1970s collections were not available to formulate other metrics.

Four Mattawoman Creek mainstem stations sampled in 1971 by O'Dell et al. (1975) were sampled by Hall et al. (1992) during 1989-1991 for water quality and ichthyoplankton. Count data were available for 1991 in a tabular summary at the sample level and these data were converted to presence-absence. Hall et al. (1992) collected ichthyoplankton with 0.5 m diameter plankton nets (3:1 length to opening ratio and 363μ mesh set for 2-5 minutes, depending on flow) suspended in the stream channel between two posts instead of stream drift nets. Changes in spawning site occupation among the current study (2008-2013), 1971 (O'Dell et al. 1975) and 1991 (Hall et al. 1992) were compared to C/ha in Mattawoman Creek. Historical and recent C/ha were compared to site occupation for Piscataway Creek (1971, 2008-2009, and 2012 -2013), Bush River (1973; O'Dell et al. 1975) and 2005-2008 (McGinty et al. 2009; Uphoff et al. 2010), and Deer Creek (1972; O'Dell et al. 1975) and 2012-2013.

The proportion of samples where Herring eggs and-or larvae were present (P_{herr}) was estimated for Matta-

RKM	12.4	18.1	27	30	34.9	38.8
Months	1 to 12	4 to 9	4 to 9	8,9	4 to 9	8,9
Sum	218	8	9	2	9	2
			Years Sampled			
1970			70	70	70	70
1971	71					
1974	74	74	74		74	
1975	75					
1976	76					
1977	77					
1978	78					
1979	79					
1980	80					
1981	81					
1982	82					
1983	83					
1984	84					
1985	85					
1986	86					
1987	87					
1988	88					
1989	89					

Table 1-2. Summary of historical conductivity sampling in non-tidal Mattawoman Creek. RKM = site location in river km from mouth; Months = months when samples were drawn; Sum = sum of samples for all years.

woman Creek mainstem stations (MC1-MC4) during 1991 and 2008-2013. Volunteer sampling of ichthyoplankton in Piscataway Creek (2008-2009 and 2012-2013), Bush River (2005-2008; McGinty et al. 2009), and Deer Creek (2012-2013) also provided sufficient sample sizes to estimate P_{herr} for those locations and years. Herring was the only species group represented with adequate sample sizes for reasonably precise annual estimates. Mainstem stations (PC1-PC3) and Tinkers Creek (PTC1) were used in Piscataway Creek. Streams that were sampled in all years in Bush River were analyzed. Deer Creek stations SU01, SU04, and SU05 correspond to O'Dell et al. (1975) sites 1, 2, and 3 respectively. Two additional sites, SU02 and SU03, were sampled in this system as well.

For the stations within the rivers described above, the proportion of samples with Herring eggs and-or larvae present (P_{herr}) was estimated as

$$N_{present} / N_{total};$$

where $N_{present}$ equaled the number of samples with Herring eggs and-or larvae present and N_{total} equaled the total number of samples taken. The SD of P_{herr} was estimated as

$$SD = [(P_{herr}\ (1- P_{herr}))\ /\ N_{total}]^{0.5} \text{ (Ott 1977).}$$

The 90% confidence intervals were constructed as

$$P_{herr} \pm (1.44\ SD).$$

White Perch and Yellow Perch have been present in samples at the downstream-most one or two stations during 1989-1991 (Hall et al. 1992) and 2008-2013 in Mattawoman Creek. We pooled three years (1989-1991, 2008-2010, and 2011-2013) to estimate the proportion of samples with White or Yellow Perch eggs and larvae in order to gain enough precision to separate these estimates from zero. Formulae for estimating proportions were the same as for estimating P_{herr} and its SD and 90% CI's (see above). White Perch spawning occurred at MC1 and MC2. Yellow Perch spawning was only detected at Station MC1.

Regression analyses examined relationships of development (C/ha) with standardized conductivity measurements (median conductivity adjusted for Coastal Plain or Piedmont background level; see below), C/ha and Herring spawning intensity (P_{herr}), and standardized conductivity with P_{herr}. Data were from Bush River and Mattawoman, Piscataway, and Deer Creeks. Seventeen estimates of C/ha and P_{herr} were available (1991 estimates for Mattawoman Creek could be included), while sixteen estimates were available for standardized conductivity (Mattawoman Creek data were not available for 1991). Examination of scatter plots suggested that a linear relationship was the obvious choice for C/ha and P_{herr}, but that either linear or curvilinear relationships might be applicable to C/ha with standardized conductivity and standardized conductivity with P_{herr}. Power functions were used to fit curvilinear models:

$$Y = a \cdot X^b$$

where Y = dependent variable (standardized conductivity or P_{herr}), X = independent variable (standardized conductivity or C/ha), a is a scaling coefficient and b is a shape parameter. Linear regressions were analyzed in Excel, while the non-linear regression analysis used Proc NLIN (Freund and Littell 2006). A linear or nonlinear model was considered the best descriptor if it was significant at $\alpha \leq 0.05$ (both were two parameter models), it explained more variability than the other (r^2 for linear and approximate r^2 for nonlinear) and examination of residuals did not suggest a pattern. We expected negative relationships of P_{herr} with C/ha and standardized conductivity, while standardized conductivity and C/ha were expected to be positively related.

Conductivity was summarized as the median for the same stations that were used to estimate P_{herr} and was standardized by dividing by an estimate of the background expected from a stream absent anthropogenic influence (Morgan et al. 2012; see below). Piedmont and Coastal Plain streams in Maryland have different background levels of conductivity (Morgan et al. 2012). Morgan et al. (2012) provided two sets of methods of estimating spring base flow background conductivity for two different sets of Maryland ecoregions, for a total set of four potential background estimates. We chose the option featuring Maryland Biological Stream Survey (MBSS) Coastal Plain and Piedmont regions and the 25th percentile background level for conductivity. These regions had larger sample sizes than the other options and background conductivity in the Coastal Plain fell much closer to the observed range estimated for Mattawoman Creek in 1991 (61-114 µS/cm) when development was relatively low (Hall et al. 1992). Background conductivity used to adjust median conductivities was 109 µS/cm in Coastal Plain streams and 150 µS/cm in Piedmont streams.

Results

Development level of the watersheds of Piscataway, Mattawoman, and Deer Creeks and Bush River started at approximately 0.05 C/ha in 1950, (Figure 1-6). Surveys conducted by O'Dell et al. (1975, 1980) in the 1970s, sampled largely rural watersheds (C/ha ≤ 0.27) except for Piscataway Creek (C/ha = 0.47). By 1991, C/ha in Mattawoman Creek was similar to that of Piscataway in 1971. By the mid-2000s Bush River and Piscataway Creek were at higher suburban levels of development (~1.30 C/ha) than Mattawoman Creek (~0.80 C/ha). Deer Creek, zoned for agriculture and preservation, remained rural through 2013 (0.24 C/ha; Figure 1-6).

In 2013, conductivity in mainstem Mattawoman Creek was steady throughout the sampling period and was slightly higher than the 1991 maximum (114 µS/cm; Figure 1-7). Four of 12 measurements at MC1 and one measurement each at MC2 and MC3 (April 14 and May 12, respectively) fell below the 1991 maximum. Conductivity in the tributaries MUT 3-5 all fell within or below the range reported by Hall et al. (1992) for the mainstem. This general pattern has held for years that conductivity has been monitored. Conductivities in Mattawoman Creek's mainstem stations in 2009 were highly elevated in early March following application of road salt in response to a significant snowfall that occurred just prior to the start of the survey (Uphoff et al. 2010). Measurements during 2009

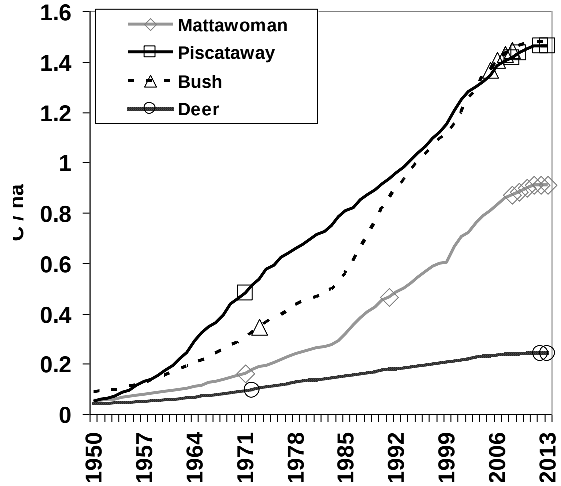

Figure 1-6. Trends in counts of structures per hectare (C/ha) during 1950-2011 in Piscataway Creek, Mattawoman Creek, Deer Creek, and Bush River watersheds. Updates estimates of C/ha were not available for 2012 or 2013. Large symbols indicate years when stream ichthyoplankton was sampled.

steadily declined for nearly a month before leveling off slightly above the 1989-1991 maximum. There was a general pattern among years of higher conductivity at the most upstream mainstem site (MC4) followed by declining conductivity downstream to the site on the tidal border. This pattern and low conductivities at the unnamed tributaries indicated that development at and above MC4 was affecting water quality (Figure 1-7).

Conductivity levels in Piscataway Creek and Bush River were elevated when compared to Mattawoman Creek (Table 1-3). With the exception of Piscataway Creek in 2012 (median = 195 µS/cm), median conductivity estimates during spawning surveys were always greater than 200 µS/cm in Piscataway Creek and Bush River during the 2000s. Median conductivity in Mattawoman Creek was in excess of 200 µS/cm during 2009 and was less than 155 µS/cm during the remaining five years (Table 1-3).

During 1970-1989, 73% of monthly median conductivity estimates in Mattawoman Creek were at or below the background level for Coastal Plain streams; C/ha in the watershed increased from 0.25 to 0.41. Higher monthly median conductivities in the non-tidal stream were more frequent nearest the confluence with Mattawoman Creek's estuary and in the vicinity of Waldorf (RKM 35) (Figure 1-8). Conductivity medians were highly variable at the upstream station nearest Waldorf during 1970-1989. During 2008-2013 (C/ha = 0.85-0.91), median spawning survey conductivi-

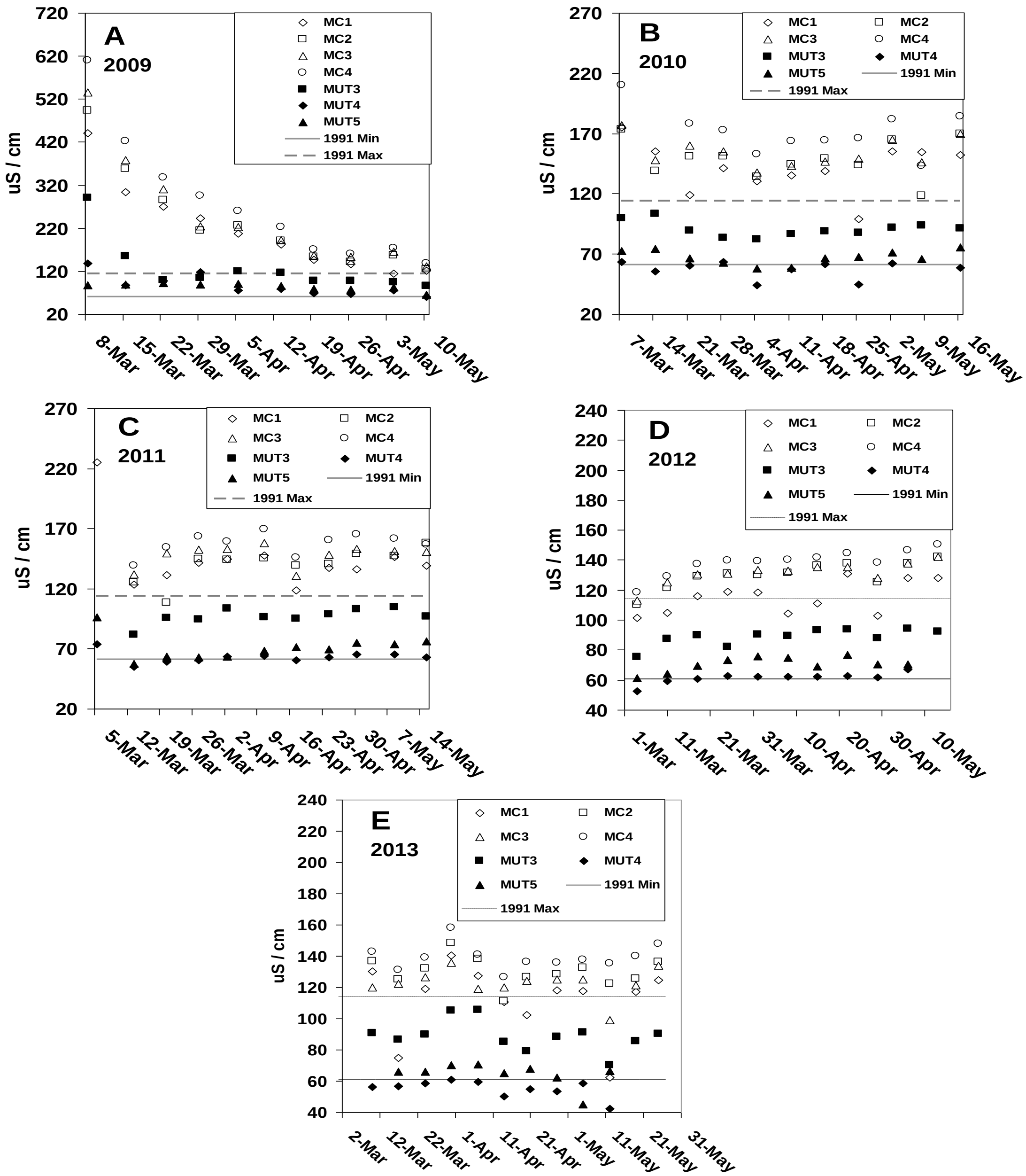

Figure 1-7. Stream conductivity measurements (µS/cm), by station and date, in Mattawoman Creek during (A) 2009, (B) 2010, (C) 2011, (D) 2012, and (E) 2013. Lines indicate conductivity range measured at mainstem sites (MC1–MC4) during 1991 by Hall et al. (1992).

ties at mainstem stations MC2 to MC4, above the confluence of Mattawoman Creek's stream and estuary (MC1), were elevated beyond nearly all 1979-1989 monthly medians and increased with upstream distance toward Waldorf. Most measurements at MC1 fell within the upper half of the range observed during 1970-1989 (Figure 1-8). None of the non-tidal conductivity medians estimated at any site during 2008-

Conductivity	Year								
	2005	2006	2007	2008	2009	2010	2011	2012	2013
					Mattawoman				
Mean				120.1	244.5	153.7	147.5	128.9	126.1
Standard Error				3.8	19.2	38	2.8	1.9	2.4
Median				124.6	211	152.3	147.3	130.9	126.5
Kurtosis				2.1	1.41	1.3	8.29	-0.26	5.01
Skewness				-1.41	1.37	0.03	1.72	-0.67	-1.70
Range				102	495	111	117	49	96
Minimum				47	115	99	109	102	63
Maximum				148.2	610	210	225	151	158
Count				39	40	43	44	44	48
					Bush				
Mean	269	206	263	237					
Standard Error	25	5	16	6					
Median	230	208	219	234					
Kurtosis	38	2	22	7					
Skewness	6	-1	4	0					
Range	1861	321	1083	425					
Minimum	79	0	105	10					
Maximum	1940	321	1187	435					
Count	81	106	79	77					
					Piscataway				
Mean				218.4	305.4			211.4	245
Standard Error				7.4	19.4			5.9	6.9
Median				210.4	260.6			195.1	238.4
Kurtosis				-0.38	1.85			0.11	-0.29
Skewness				0.75	1.32			0.92	0.73
Range				138	641			163	173
Minimum				163	97			145	181
Maximum				301	737			308	354
Count				29	50			44	44
					Deer				
Mean								174.9	175.6
Standard Error								1.02	1.5
Median								176.8	177.7
Kurtosis								17.22	13.88
Skewness								-3.78	-2.25
Range								39.3	122
Minimum								140.2	93
Maximum								179.5	215
Count								44	87

Table 1-3. Summary statistics of conductivity (µS/cm) for mainstem stations in Piscataway, Mattawoman and Deer Creeks, and Bush River during 2005-2013. Unnamed tributaries were excluded from analysis. Tinkers Creek was included with mainstem stations in Piscataway Creek.

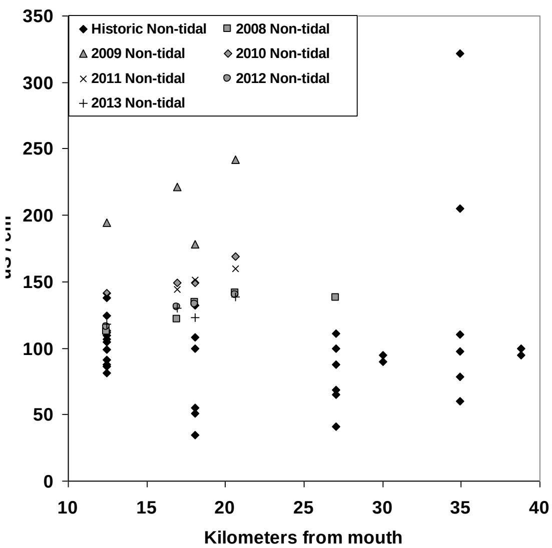

Figure 1-8. Historical (1970-1989) median conductivity measurements and current (2008-2013) anadromous spawning survey median conductivity in non-tidal Mattawoman Creek (between the junction with the subestuary and Waldorf) plotted against distance from the mouth. The two stations furthest upstream are nearest Waldorf. Median conductivity was measured during March-May, 2008-2013, and varying time periods (see Table 1-2) during 1970-1989.

2013 were at or below the Coastal Plain stream background criterion.

Quality assurance vials only contained additional eggs and-or larvae of target species already present in the original vials. No new target species were detected during the assessment of the QA vials.

Anadromous fish spawning site occupation in fluvial Mattawoman Creek improved during 2008-2013 but was less consistent than during 1971 and 1989-1991 (historical spawning period; Table 1-4). Herring spawning was detected during 2008-2013 at historical mainstem stations. Herring spawning was absent at stations MC2, MC4, and MUT3 during 2008-2009. Site occupation has increased since 2009 and all four mainstem stations had Herring eggs and-or larvae during 2010-2013. Herring spawning was detected at MUT3 in 2011-2013 and at MUT4 in 2012. Herring spawning was detected at all mainstem stations in 1971 and 1991. Stream spawning of White Perch in Mattawoman Creek was not detected during 2009, 2011, and 2012, but spawning was detected at MC1 during 2008 and 2010, and at MC1 and MC2 during 2013. During 1971 and 1989-1991, White Perch spawning occurred annually at MC1 and intermittently at MC2; these two stations were represented every year. Prior to 2008-2013, MC3 was sampled in 1971 and 1991 and White Perch were only present during 1971. Yellow Perch spawning occurred at station MC1 every year except 2009 and 2012. Station MC1

	Year									
Station	1971	1989	1990	1991	2008	2009	2010	2011	2012	2013
					Herring					
MC1	1	1	1	1	1	1	1	1	1	1
MC2	1	1	1	1	0	0	1	1	1	1
MC3	1			1	1	1	1	1	1	1
MC4	1			1	0	0	1	1	1	1
MUT3	1				0	0	0	1	1	1
MUT4							0	0	1	0
MUT5	1				1	0	0	0	0	0
					White Perch					
MC1	1	1	1	1	1	0	1	0	0	1
MC2	0	0	1	0	0	0	0	0	0	1
MC3	1			0	0	0	0	0	0	0
					Yellow Perch					
MC1	1	1	1	1	1	0	1	1	0	1

Table 1-4. Presence-absence of Herring (Blueback Herring, Hickory and American Shad, and Alewife), White Perch, and Yellow Perch stream spawning in Mattawoman Creek during 1971, 1989-1991, and 2008-2013. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-2.

Station	Year 1971	2008	2009	2012	2013
Herring					
PC1	1	0	0	1	1
PC2	1	0	1	1	1
PC3	1	0	0	1	1
PTC1	1	0	0	1	1
PUT4	1		0	0	0
White Perch					
PC1	1	0	0	0	0
PC2	1	0	0	0	0

Table 1-5. Presence-absence of Herring (Blueback Herring, Hickory and American Shad, and Alewife) and White Perch stream spawning in Piscataway Creek during 1971, 2008-2009, and 2012-2013. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-3.

was the only stream station in Mattawoman Creek where Yellow Perch spawning has been detected in surveys conducted since 1971 (Table 1-4).

Herring spawning was detected at all mainstem sites in Piscataway Creek in 2012 and 2013. Stream spawning of anadromous fish had nearly ceased in Piscataway Creek between 1971 and 2008-2009 (Table 1-5). Herring spawning was not detected at any site in the Piscataway Creek drainage during 2008 and was only detected on one date and location (one Herring larvae on April 28 at PC2) in 2009. Stream spawning of White Perch was detected at PC1 and PC2 in 1971 but has not been detected during 2008-2009 and 2012-2013 (Table 1-5).

There was no obvious decline in Herring spawning in the Bush River stations between 1973 and 2005-2008, but occurrences of White and Yellow Perch became far less frequent (Table 1-6). During 1973, Herring spawning was detected at 7 of 12 Bush River stream sites sampled; however, during 2005-2008 Herring spawning was detected in as few as 5 of 12 sites or as many as 8 of 8 sites sampled in the Bush River. White Perch spawning in the Bush River was detected at 8 of 12 sites sampled during 1973 and at one site in one year during 2005-2008. The pattern of stream spawning site occupation of Yellow Perch in Bush River was similar to that of White Perch spawning. Yellow Perch spawned at five of 12 sites during 1973. Yellow Perch spawning was not detected during

Station	Year 1973	2005	2006	2007	2008
Herring					
BBR1	0	1	1	1	1
BBR2	0	0	0		
BCR1	1	0	0	1	0
BGR1	0	1	1	1	
BGR2	1	0	0		
BGRT					0
BHH1	0	0	1	1	1
BHHT					0
BJR1	0	1	1	1	0
BOP1	1	1	1	1	1
BSR1	1	0	0		
BWR1	1	0	0	1	0
BWR2	1	0	0		
BWRT					1
BUN1	1	1	1	1	
White Perch					
BBR1	1	0	0	0	0
BBR2	0	0	0		
BCR1	1	0	0	0	0
BGR1	1	0	0	0	
BGR2	1	0	0		
BGRT					0
BHH1	0	0	0	0	0
BHHT					0
BJR1	0	0	0	0	0
BOP1	1	0	0	1	0
BSR1	0	0	0		
BWR1	1	0	0	0	0
BWR2	1	0	0		
BWRT					0
BUN1	1	0	0	0	

Table 1-6. Presence-absence of Herring (Blueback Herring, Hickory and American Shad, and Alewife), White Perch, and Yellow Perch stream spawning in Bush River during 1973 and 2005-2008. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-4.

Station	Year				
	1973	2005	2006	2007	2008
	Yellow Perch				
BBR1	1		0		0
BBR2	1		1		
BCR1	0		0		0
BGR1	1		1		
BGR2	0	0	1	0	
BGRT					0
BHH1	0	0	0		0
BHHT					0
BJR1	1	0	0	0	0
BOP1	0	0	0	0	0
BSR1	0	0	0	0	
BWR1	1	0	1	0	0
BWR2	0	0	0		
BWRT					0
BUN1	0	0	0	0	

Table 1-6 (continued). Presence-absence of Herring (Blueback Herring, Hickory and American Shad, and Alewife), White Perch, and Yellow Perch stream spawning in Bush River during 1973 and 2005-2008. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-4.

three of four surveys during 2005-2008, but was detected at four of 12 sites during 2006 (Table 1-6).

O'Dell et al. (1975) reported Herring, White Perch, and Yellow Perch spawning in Deer Creek during 1972 (Table 1-7). Three sites were sampled during 1972 in Deer Creek and one was located upstream of an impassable dam near Darlington (a fish passage was installed there in 1999). During 1972 Herring spawning was detected at both sites below the dam (SU01 and SU03), while White and Yellow Perch spawning were detected at the mouth (SU01). During 2012-2013, Herring spawning was detected at all sites sampled in both years. White Perch spawning was not detected in Deer Creek in 2012 but was detected at three sites in 2013. Yellow Perch spawning was detected at the two stations closest to the mouth in 2012 but was not detected in 2013 (Table 1-7).

The 90% confidence intervals of P_{herr} (Figure 1-9) provided sufficient precision for us to categorize four levels of stream spawning: very low levels at or indistinguishable from zero based on confidence interval overlap (level 0); a low level of spawning that could be distinguished from zero (level 1); a mid-level of spawning that could usually be separated from the low levels (level 2); and a high level (3) of spawning likely to be higher than the mid-level. Stream spawning in Mattawoman Creek was categorized at levels 1 (2008-2009), 2 (2010 and 2012), and level 3 (1991, 2011, and 2013). Spawning in Piscataway Creek was at level 0 during 2008-2009, at level 2 during 2012, and at level 1 during 2013. Bush River spawning was characterized by levels 0 (2006) and 1 (2005 and 2007-2008). Deer Creek, with the least developed watershed, was characterized by the highest level of spawning (level 3) during 2012-2013 (Figure 1-9).

The 90% CI's of proportions of samples with White Perch eggs and larvae at stations MC1 and MC2, pooled in 3-year intervals, indicated that less spawning occurred in Mattawoman Creek during 2008-2010 than during 1989-1991 (Figure 1-10). Status of spawn-

Station	Year		
	1972	2012	2013
	Herring		
SU01	1	1	1
SU02	---	1	1
SU03	---	1	1
SU04	1	1	1
SU05	0	---	1
	White Perch		
SU01	1	0	1
SU02	---	0	1
SU03	---	0	0
SU04	0	0	1
SU05	0	---	0
	Yellow Perch		
SU01	1	1	0
SU02	---	1	0
SU03	---	0	0
SU04	0	0	0
SU05	0	---	0

Table 1-7. Presence-absence of Herring (Blueback Herring, Hickory and American Shad, and Alewife) and White Perch stream spawning in Deer Creek during 1972 and 2012-2013. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample.

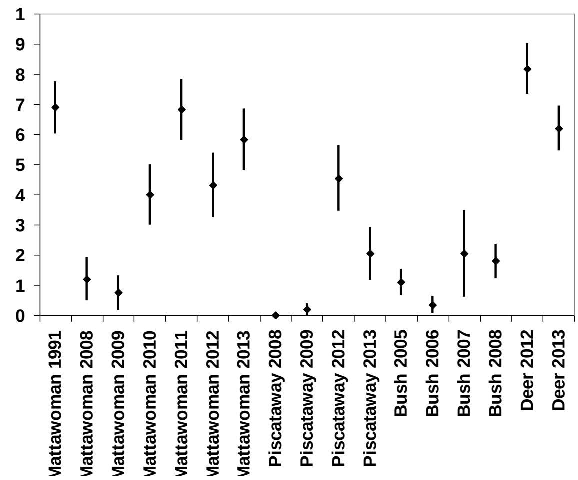

Figure 1-9. Proportion of samples (*P_{herr}*) with Herring and its 90% confidence interval for stream ichthyoplankton surveys in Mattawoman, Piscataway and Deer Creeks, and Bush River.

ing during 2011-2013 was not clear since 90% CI's of the proportion of samples with White Perch eggs and larvae during 2011-2013 overlapped both 1989-1991 and 2008-2010. The 90% CI's for stream spawning of Yellow Perch (at MC1 only) overlapped for 1989-1991, 2008-2010, and 2011-2013, indicating significant change had not occurred (Figure 1-10).

Standardized conductivity increased with development, while P_{herr} declined with both development and standardized conductivity. Regression analyses indicated significant and logical relationships among P_{herr}, C/ha, and standardized median conductivity (Table 1-8). The relationship of C/ha with standardized median conductivity was linear, significant, and positive (r^2 = 0.38, P = 0.01, N = 16; Figure 1-11). Estimates of P_{herr} were linearly, significantly, and negatively related to C/ha (r^2 = 0.57, P < 0.0001, N = 17). A negative curvilinear regression best described the relationship of

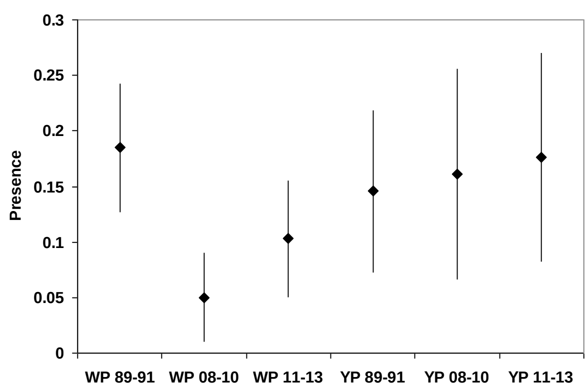

Figure 1-10. Mattawoman data pooled across years to form estimates of proportions of samples with White Perch (WP) eggs and-or larvae (sites MC1 and MC2) or Yellow Perch (YP) eggs and-or larvae (MC1) for 1989-1991 collections compared to 2008-2010 and 2011-2013 collections at the same combination of sites.

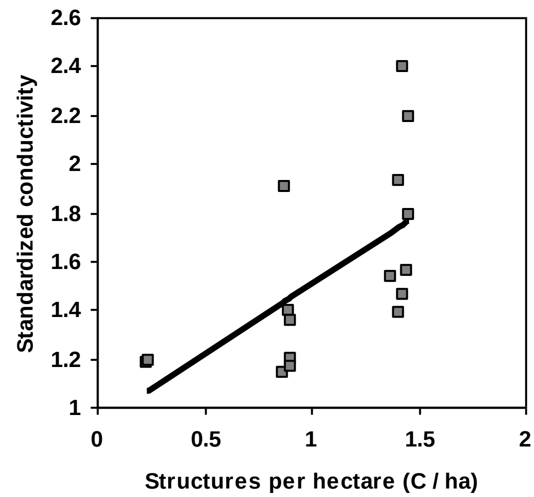

Figure 1-11. Standardized median conductivity during spring spawning surveys and level of development (C/ha). Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates in Morgan et al. (2012).

P_{herr} and standardized median conductivity (approximate r^2 = 0.37 P < 0.0001, N = 16; Figure 1-12). Low estimates of P_{herr} were much more frequent beyond the C/ha threshold (0.83 C/ha) or when standardized conductivity was 1.5-times or more than the baseline level (Figure 1-12).

Discussion

Proportion of samples with Herring eggs and-or larvae (P_{herr}) provided a reasonably precise estimate of relative abundance based on encounter rate. Regression analyses indicated significant and logical relationships among P_{herr}, C/ha, and conductivity consistent with the hypothesis that urbanization was detrimental to stream spawning. Conductivity was positively related with C/ha in our analysis and with urbanization in other studies (Wang and Yin 1997; Paul and Meyer 2001; Wenner et al. 2003; Morgan et al. 2007; Carlisle et al. 2010; Morgan et al. 2012). Limburg and Schmidt

Linear Model: Structure density (C / ha) = standardized conductivity

$r^2 = 0.38$

ANOVA	*df*	*SS*	*MS*	*F*	*Significance F*
Regression	1	0.8553457	0.8553457	8.616	0.011
Residual	14	1.3898789	0.0992771		
Total	15	2.2452246			

	Estimate	*SE*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*
Intercept	0.93	0.23	4.11	0.0010672	0.44	1.41
C/ha	0.57	0.2	2.94	0.0108581	0.15	0.99

Linear Model: Structure density (C / ha) = Proportion of samples with herring eggs and larvae (P_{herr})

$r^2 = 0.57$

ANOVA	*df*	*SS*	*MS*	*F*	*Significance F*
Regression	1	0.6588704	0.6588704	19.759	0
Residual	15	0.5001913	0.0333461		
Total	16	1.1590618			

	Estimate	*SE*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*
Intercept	0.82	0.12	6.88	5.20E-06	0.57	1.08
c/ha	-0.47	0.11	-4.45	0.0004724	-0.7	-0.25

Nonlinear Model: Standardized conductivity = Proportion of samples with herring eggs and larvae (P_{herr})

Approximate r^2 =
0.37

Source	*DF*	*Sum of Squares*	*Mean Square*	*F Value*	*Approximate Pr > F*
Model	2	1.8877	0.9438	20.75	<0.0001
Error	14	0.6367	0.0455		
Uncorrected Total	16	2.5244			

Parameter	*Estimate*	*Approximate SE*	*Lower 95%*	*Upper 95%*
a	0.76	0.24	0.24	1.28
b	-2.59	1.18	-5.12	-0.06

Table 1-8. Summary of best regression models for standardized conductivity (annual median/province background) versus development level (C/ha), proportion of samples with herring eggs or larvae (P_{herr}) versus C/ha, and P_{herr} versus standardized conductivity.

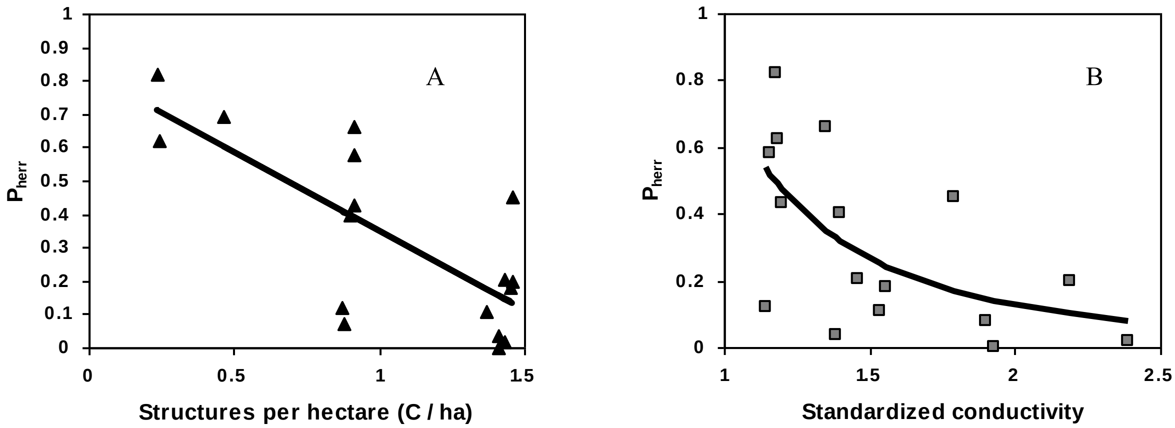

Figure 1-12. (A) Proportion of stream samples with Herring eggs or larvae (P_{herr}) and level of development (C/ha). (B) P_{herr} and standardized median spawning survey conductivity. Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates in Morgan et al. (2012).

(1990) found a highly nonlinear relationship of densities of anadromous fish (mostly Alewife) eggs and larvae to urbanization in Hudson River tributaries with a strong negative threshold at low levels of development.

An unavoidable assumption of regression analyses of P_{herr}, C/ha, and summarized conductivity was that watersheds at different levels of development were a substitute for time-series. Extended time-series of watershed specific data were not available. Mixing physiographic provinces in this analysis had the potential to increase scatter of points, but standardizing median conductivity to background conductivity moderated the province effect in analyses with that variable. Differential changes in physical stream habitat and flow due to differences in geographic provinces could also have affected fits of regressions. Estimates of C/ha would have indexed these physical changes as well as water chemistry, while standardized conductivity would only have represented changes in water chemistry. Estimates of C/ha explained more variation in P_{herr} (57%) than standardized conductivity (37%).

Elevated conductivity, related primarily to chloride from road salt (but including most inorganic acids and bases; APHA 1979), has emerged as an indicator of watershed development (Wenner et al. 2003; Kaushal 2005; Morgan et al. 2007; Morgan et al. 2012). Use of salt as a deicer may lead to both “shock loads” of salt that may be acutely toxic to freshwater biota and elevated baselines (increased average concentrations) of chloride that have been associated with decreased fish and benthic diversity (Kaushal et al. 2005; Wheeler et al. 2005; Morgan et al. 2007; 2012). Commonly used anti-clumping agents for road salt (ferro- and ferricyanide) that are not thought to be directly toxic are of concern because they can break down into toxic cyanide under exposure to ultraviolet light. Although the degree of breakdown into cyanide in nature is unclear (Pablo et al. 1996; Transportation Research Board 2007), these compounds have been implicated in fish kills (Burdick and Lipschuetz 1950; Pablo et al. 1996; Transportation Research Board 2007). Heavy metals and phosphorous may also be associated with road salt (Transportation Research Board 2007).

At least two hypotheses can be formed to relate decreased anadromous fish spawning to conductivity and road salt use. First, eggs and larvae may die in response to sudden changes in salinity and potentially toxic amounts of associated contaminants and additives. Second, changing stream chemistry may cause disorientation and disrupted upstream migration.

Levels of salinity associated with our conductivity measurements are very low (maximum 0.1 ppt) and anadromous fish spawn successfully in brackish water (Klauda et al. 1991; Piavis et al. 1991; Setzler-Hamilton 1991). A rapid increase might result in osmotic stress and lower survival since salinity represents osmotic cost for fish eggs and larvae (Research Council of Norway 2009).

Elevated stream conductivity may prevent anadromous fish from recognizing and ascending streams. Alewife and Herring are thought to home to natal rivers to spawn (ASMFC 2009a; ASMFC 2009b), while Yellow and White Perch populations are generally tributary-specific (Setzler-Hamilton 1991; Yellow Perch Workgroup 2002). Physiological details of spawning migration are not well described for our target species, but homing migrations in anadromous American Shad and Salmon have been connected with chemical composition, smell, and pH of spawning streams (Royce-Malmgren and Watson 1987; Dittman and Quinn 1996; Carruth et al. 2002; Leggett 2004). Conductivity is related to total dissolved solids in water (Cole 1975) which reflects chemical composition.

Processes such as flooding, riverbank erosion, and landslides vary by geographic province (Cleaves 2003) and influence physical characteristics of streams. Unconsolidated sediments (layers of sand, silt, and clay) underlie the Coastal Plain and broad plains of low relief and wetlands characterize the terrain (Cleaves 2003). Coastal Plain streams have low flows and sand or gravel bottoms (Boward et al. 1999). The Piedmont is underlain by metamorphic rocks and characterized by narrow valleys and steep slopes, with regions of higher land between streams in the same drainage. Most Piedmont streams are of moderate slope with rock or bedrock bottoms (Boward et al. 1999). The Piedmont is an area of higher gradient change and more diverse and larger substrates than the Coastal Plain (Harris and Hightower 2011) that may offer greater variety of Herring spawning habitats.

Urbanization and physiographic province both affect discharge and sediment supply of streams (Paul and Meyer 2001; Cleaves 2003) that, in turn, could affect location, substrate composition, extent and success of spawning. Alewife spawn in sluggish flows, while Blueback Herring spawn in sluggish to swift flows (Pardue 1983). American Shad select spawning habitat based on macrohabitat features (Harris and Hightower 2011) and spawn in moderate to swift flows (Hightower and Sparks 2003). Spawning substrates for Herring include gravel, sand, and detritus (Pardue 1983). Detritus loads in subestuaries are strongly associated with development (see Section 1-3) and urbanization affects the quality and quantity of organic matter in streams (Paul and Meyer 2001) that feed into subestuaries.

Herring spawning became more variable in streams as watersheds developed. The surveys from watersheds with C/ha of 0.46 or less had high P_{herr}. Estimates of P_{herr} from Mattawoman Creek during 2008-2013 (C/ha was 0.85-0.91) varied from barely different from zero to high. Eggs and larvae were nearly absent from fluvial Piscataway Creek during 2008-2009, but P_{herr} rebounded to 0.45 in 2012 and then dropped again to 0.2 in 2013 (C/ha was 1.39-1.46). The rebound in Herring spawning in Piscataway Creek during 2012 was concurrent with the lowest mean and median conductivities encountered there in the four years sampled. Variability of Herring spawning in Bush River during 2005-2008 involved "colonization" of new sites as well as absence from sites of historical spawning.

Magnitude of P_{herr} may indicate how much habitat is available or how attractive it is from year to year more-so than abundance of spawners. In developed watersheds, a combination of urban and natural stream processes may create varying amounts of ephemeral spawning habitat annually and dampen spawning migrations through increased conductivity. Observed variation in P_{herr} would indicate wide annual and regional fluctuations in population size. However, stock assessments of Alewife and Blueback Herring along the Atlantic coast, including those in Maryland, indicate they are in decline or are at depressed stable levels (ASMFC 2009a; 2009b; Limburg and Waldman 2009; Maryland Fisheries Service 2012) rather than fluctuating.

Application of presence-absence data in management needs to consider whether absence reflects a disappearance from suitable habitat or whether habitat sampled is not really habitat for the species in question (MacKenzie 2005). Our site occupation comparisons were based on the assumption that spawning sites detected in the 1970s were indicative of the extent of habitat. O'Dell et al. (1975) summarized spawning activity as the presence of any species group's egg, larva, or adult (latter from wire trap sampling) at a site and we used this criterion (spawning detected at a site or not) for a set of comparisons. Raw data for the 1970s were not available to formulate other metrics. This approach represented a presence-absence design with limited ability to detect population changes or conclude an absence of change since only a small number of sites could be sampled (limited by road crossings) and the positive statistical effect of repeated visits (Strayer 1999) was lost by summarizing all samples into a single record of occurrence in a sampling season. A single year's record was available for each of the watersheds in the 1970s and we were left assuming this distribution applied over multiple years of low development. Site occupation in Mattawoman Creek changed little, if at all, between 1971 and 1989-1991 when development was below threshold level; this represents the only data set available for this comparison.

Loss of Yellow Perch stream spawning sites coincided with increased development. When watershed development was above the threshold (C/ha $\geq$ 0.83), Yellow Perch stream spawning was not detected in some years in Mattawoman Creek (C/ha = 0.85-0.90) and most years in Bush River. Site occupation was steady when C/ha was 0.47 or less. We can demonstrate changes in stream spawning site occupation of White Perch and Herring between the 1970s and

2000s, but are unable to conclude that development had an impact. White Perch stream spawning largely ceased in our study streams between the 1970s and the 2000s. However, it disappeared in every watershed regardless of development level, except in Aberdeen Proving Grounds where White Perch occupation was observed at three of the four historical sites sampled (McGinty et al. 2009). Herring spawning has not occurred at some sites where it was documented in the 1970s, occurred at sites where it had not been detected previously, or continued at sites where it had been detected.

Proportion of positive samples (P_{herr} for example) provided an economical and precise alternative estimate of relative abundance based on encounter rate rather than counts. Encounter rate is readily related to the probability of detecting a population (Strayer 1999). Proportions of positive or zero catch indices were found to be robust indicators of abundance of Yellowtail Snapper *Ocyurus chrysurus* (Bannerot and Austin 1983), age-0 White Sturgeon *Acipenser transmontanus* (Counihan et al. 1999), Pacific Sardine *Sardinops sagax* eggs (Mangel and Smith 1990), Chesapeake Bay Striped Bass eggs (Uphoff 1997), and Longfin Inshore Squid *Loligo pealeii* fishery performance (Lange 1991).

Unfortunately, estimating reasonably precise proportions of stream samples with White or Yellow Perch eggs annually will not be logistically feasible without major changes in sampling priorities. Estimates for Yellow or White Perch stream spawning would require more frequent sampling to obtain precision similar to that attained by P_{herr} since spawning occurred at fewer sites. Given staff and volunteer time limitations, this would not be possible within our current scope of operations. In Mattawoman Creek, it was, however, possible to pool data across years to form estimates of proportions of samples with White Perch eggs and larvae (sites MC1 and MC2) or Yellow Perch larvae (MC1) for 1989-1991 collections to compare with 2008-2013 collections at the same combinations of sites. These estimates did not indicate a loss in stream spawning in these downstream sites.

Volunteer-based sampling of stream spawning during 2005-2013 used only stream drift nets, while O'Dell et al. (1975) and Hall et al. (1992) determined spawning activity with ichthyoplankton nets and wire traps for adults. Tabular summaries of egg, larval, and adult catches in Hall et al. (1992) allowed for a comparison of how site use in Mattawoman Creek might have varied in 1991 with and without adult wire trap sampling. Sites estimated when eggs and-or larvae were present in one or more samples were identical to those when adults present in wire traps were included with the ichthyoplankton data (Hall et al. 1992). Similar results were obtained from the Bush River during 2006 at sites where ichthyoplankton drift nets and wire traps were used; adults were captured by traps at one site and eggs and-or larvae at nine sites with ichthyoplankton nets (Uphoff et al. 2007). Wire traps set in the Bush River during 2007 did not indicate different results than ichthyoplankton sampling for Herring and Yellow Perch, but White Perch adults were observed in two trap samples and not in plankton drift nets (Uphoff et al. 2008). These comparisons of trap and ichthyoplankton sampling indicated it was unlikely that an absence of adult wire trap sampling would impact interpretation of spawning sites when multiple years of data were available.

The different method used to collect ichthyoplankton in Mattawoman Creek during 1991 could bias that estimate of P_{herr}, although presence-absence data tend to be robust to errors and biases in sampling (Green 1979). Removal of 1991 data lowered the fit between C/ha and P_{herr} (from $r^2 = 0.57$, P = 0.0005 to $r^2 = 0.52$, P = 0.002), but did not alter the negative relationship (95% CI's of slopes and intercepts of both models overlapped).

Absence of detectable stream spawning does not necessarily indicate an absence of spawning in the estuarine portion of these systems. Estuarine Yellow Perch presence-absence surveys in Mattawoman and Piscataway Creeks, and Bush River did not indicate that lack of detectable stream spawning corresponded to their elimination from these subestuaries. Yellow Perch larvae were present in upper reaches of both subestuaries (see Section 2). Yellow Perch do not appear to be dependent on non-tidal stream spawning, but their use may confer benefit to the population through expanded spawning habitat diversity. Stream spawning is very important to Yellow Perch anglers since it provides access for shore fisherman and most recreational harvest probably occurs during spawning season (Yellow Perch Workgroup 2002).

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Job 1, Section 2: Estuarine Yellow Perch Presence-Absence Sampling

Introduction

Presence-absence sampling for Yellow Perch larvae was conducted in the upper tidal reaches of the Nanticoke, Northeast, Choptank, Patuxent, and Bush Rivers and Mattawoman, Nanjemoy, and Piscataway Creeks during late March through April, 2013 (Figure 2-1). Annual L_p, the proportion of tows with Yellow Perch larvae during a standard time period and where larvae would be expected, provides a cost-effective measure of the product of egg production and survival through the early post-larval stage. In 2013 we continued examining relationships of L_p with estimates of development and other land uses.

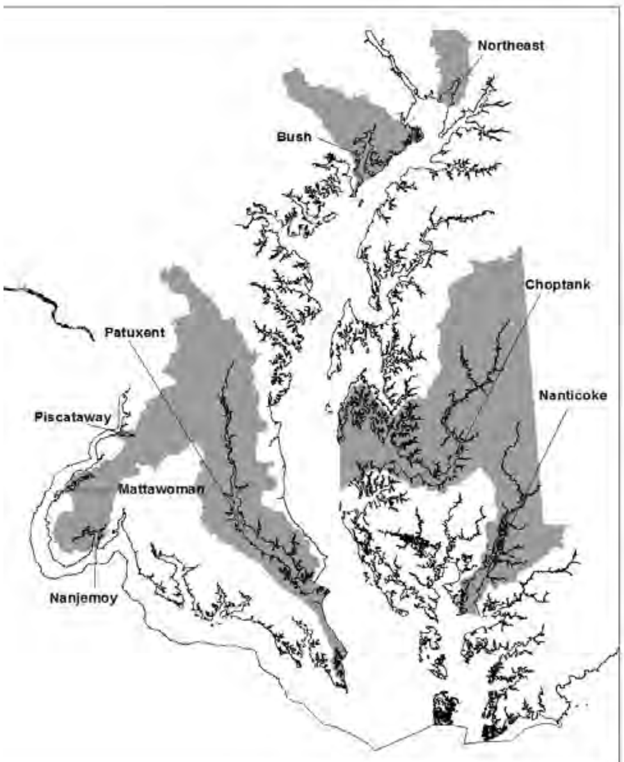

Figure 2-1. Sampling areas for the 2013 Yellow Perch larval presence absence study. Nanticoke River watershed delineation was unavailable for Delaware and Northeast and was unavailable for Pennsylvania.

We examined a hypothesis that development negatively influenced watershed organic matter (OM) dynamics, altering zooplankton production important for Yellow Perch larval feeding success and survival (the OM hypothesis) using the empirical-statistical approach recommended by Austin and Ingham (1978) and Crecco and Savoy (1984) for resolving the effects of environment on fish recruitment. This approach offers a working hypothesis that is tested for validity with empirical data and a thorough statistical analysis.

Years of high spring discharge favor anadromous fish recruitment in Chesapeake Bay (Hoffman et al. 2007; Martino and Houde 2010) and may represent episodes of hydrologic transport of accumulated OM from watersheds that fuel zooplankton production and feeding success (McClain et al. 2003). Under natural conditions, riparian marshes and forests would provide OM subsidies in high discharge years (Hoffman et al. 2007), while phytoplankton would be the primary source of OM in years of lesser flow. Shortage of appropriate food has been frequently hypothesized to cause high mortality of fish larvae (Martin et al. 1985; Miller et al. 1988; Heath1992).

Urbanization was expected to negatively impact Yellow Perch larval feeding success because it affects the quality and quantity of OM in streams (Paul and Meyer 2001) and was negatively associated with extent of wetlands in many subestuary watersheds evaluated in this study (Uphoff et al. 2011a). Riparian zones and floodplains that are sources of OM become disconnected from stream channels by stormwater management in suburban and urban watersheds (Craig et al. 2008; Kaushal et al. 2008; Elmore and Kaushal 2008; Brush 2009; NRC 2009), altering quantity and transport of OM (Paul and Meyer 2001; McClain et al. 2003; Stanley et al. 2012).

Correlation analyses examined associations of C/ha and 2010-2013 feeding success, L_p, larval TL, diet composition, and relative detritus levels collected during spring surveys. Larval fish size was included because it can be critical to larval feeding and starvation (Miller et al. 1988). Uphoff et al. (2012) included factors in addition to C/ha in analyses of 2010-2011 feeding success: relative amounts of OM, larval TL, mean water temperature, and mean conductivity in analyses of feeding success. Organic matter and larval length were found to be significant influences on feeding success, but water temperature and mean conductivity were not. Analyses of 2010-2013 feeding data in this report concentrated on variables that were significant in Uphoff et al. (2012).

During 2012-2013, Yellow Perch were also collected for analysis of the ratio of ribonucleic acid (RNA) concentration to deoxyribonucleic acid (DNA) concentration in body tissue in addition to estimating L_p and feeding success. The quantity of DNA within a cell is constant within a species while the quantity of RNA varies with protein synthesis (Tardiff et al. 2005). Since growth is a function of protein synthesis, RNA/DNA ratios provide a sensitive indicator of recent growth at any given time (Buckley 1984). This ratio is a useful indicator of nutritional status and somatic growth in larval fish (Buckley 1984) that provides a method for examining connections of feeding success and larval condition (Buckley 1984; Martin et al. 1985; Wright and Martin 1985; Clemmesen 1994; Blom et al. 1997) without requiring extensive sampling and sample processing needed to measure mortality directly. Tardif et al. (2005) used RNA/DNA ratios of Yellow Perch larvae and juveniles to determine differences in productivity of managed and natural wetlands of Lake St. Pierre, Canada.

Samples were gathered from three adjacent Potomac River subestuaries exhibiting a gradient of development from rural to suburban (C/ha ranged from 0.09 to 1.46) during 2013. We expected RNA/DNA ratios to decline with increased development.

Methods

Conical plankton nets were towed from boats in upper portions of subestuaries to collect Yellow Perch larvae. Nets were 0.5-m in diameter, 1.0-m long, and constructed of 0.5 mm mesh. Nets were towed with the current for two minutes at a speed that maintained the net near the surface (approximately 2.8 km per hour). Temperature, dissolved oxygen, conductivity, and salinity were measured at each site on each sample date.

Ten sites were sampled in Nanjemoy and Mattawoman Creeks, and in Choptank, Bush, Northeast, and Nanticoke Rivers (Figure 2-1). Seven sites were sampled in Piscataway Creek. Five to ten stations were sampled on the Patuxent River. All subestuaries were sampled twice per week, although sampling in the Patuxent was not consistent and did not always follow this schedule. Larval sampling occurred during late March through mid-to-late April. Boundaries of areas sampled were determined from Yellow Perch larval presence in estuarine surveys conducted during the 1970s and 1980s (O'Dell 1987). Sites in all subestuaries (except the Nanticoke and Choptank rivers) were sampled with little spacing between tows because their larval nurseries were small. Three upstream sites in Piscataway Creek could not be sampled at very low tides.

Each sample was emptied into a glass jar and checked for larvae. Yellow Perch larvae can be readily identified in the field since they are larger and more developed than *Morone* larvae with which they could be confused (Lippson and Moran 1974).

Contents of the jar were allowed to settle and then the amount of settled OM was assigned a rank: 0=clear to not enough to define a layer; 1=defined layer on bottom; 2 = more than defined layer and up to ¼ full; 3 =more than ¼ to ½ and; 4 = more than ½ full.

If a jar contained enough OM to obscure seeing larvae, it was emptied into a pan with a dark background and observed through a 5X magnifying lens. Organic matter was moved with a probe or forceps to free larvae for observation. If OM loads, wave action, or collector uncertainty prevented positive identification, samples were preserved and taken back to the lab for sorting.

Nanjemoy, Piscataway, and Mattawoman Creeks, and Choptank River, were sampled by program personnel. Nanticoke and Northeast Rivers were voluntarily sampled by other Maryland Fisheries Service projects without charge to this grant. Patuxent and Bush Rivers were sampled by staff from the Chesapeake Bay National Estuarine Research Reserve Program and volunteers trained by our program biologists.

Composite samples of larvae were collected for feeding analyses from several sites in Piscataway, Mattawoman, and Nanjemoy Creeks, and Choptank, Nanticoke, Bush, and Northeast Rivers during several sample trips. Subsamples of postlarvae 12 mm TL or less were examined for gut contents from each day's samples of each subestuary. These larvae represented first-feeding and early postlarvae, larvae that absorbed their yolk and began active feeding (Hardy 1978). Larvae were measured to the nearest millimeter. Gut fullness was judged visually and assigned a rank: 0 = empty; 1 = up to ¼ full; 2 = up to ½ full; 3 = up to ¾ full; and 4 = full. Major food items were classified as copepods, cladocerans, or other and the presence (coded 1) or absence (coded 0) of each group was noted.

The proportion of tows with Yellow Perch larvae (L_p) was determined annually for dates spanning the first catch through the last date that larvae were consistently present as:

$$^{(1)}L_p = N_{present} / N_{total};$$

where $N_{present}$ equaled the number of samples with Yellow Perch larvae present and N_{total} equaled the total number of samples. The SD of L_p was estimated as;

$$^{(2)}\mathrm{SD} = [(L_p\,(1- L_p)) / \mathrm{N}_{\mathrm{total}}]^{0.5} \text{ (Ott 1977).}$$

The 95% confidence intervals were constructed as:

$$^{(3)}L_p \pm 1.96 \text{ SD; (Ott 1977).}$$

In general, sampling to determine L_p began during the last days of March or first days of April and ended after larvae were absent (or nearly so) for two consecutive sampling rounds. In years where larvae disappeared quickly, sampling rounds into the third week of April were included in analysis even if larvae were not collected. This sampling schedule has been maintained for tributaries sampled by program personnel since 2006. Sampling by other Fisheries Service projects and volunteers sometimes did not adhere as strictly to this schedule.

Uphoff et al. (2012) developed L_p thresholds for brackish and tidal-fresh systems. Three brackish subestuaries with C/ha $\geq$ 1.59 (10 estimates from Severn, South, and Magothy Rivers) exhibited chronically depressed L_p and their maximum L_p (0.33) was chosen as a threshold indicating serious deterioration of brackish subestuary larval nursery habitat. Similarly, tidal-fresh Piscataway Creek's four estimates of L_p (2008-2011) consistently ranked low when compared to other tidal-fresh subestuaries sampled (13th to 17th out of 17 estimates). The maximum for Piscataway Creek's four estimates, L_p = 0.65, was chosen as a threshold indicating serious deterioration of tidal-fresh larval habitat. Estimates of L_p would need to be consistently at or below this level to be considered "abnormal" as opposed to occasional depressions (Uphoff et al. 2012).

We estimated the risk that L_p was below a threshold for a tidal-fresh or brackish subestuary as one minus the cumulative proportion (expressed as a percentage) of the L_p binomial distribution function equaling or exceeding the restoration criterion. This calculation was used by Uphoff (1997) to estimate the risk that the proportion of plankton tows with Striped Bass eggs was not at a restored level.

Historical collections in the Choptank and Nanticoke rivers targeted striped bass eggs and larvae (Uphoff 1997), but Yellow Perch larvae were also common (Uphoff 1991). Uphoff et al. (2005) reviewed presence-absence of Yellow Perch larvae in past Choptank and Nanticoke River collections and found that starting dates during the first week of April or early in the second week were typical and end dates occurred during the last week of April through the first week of May. Larval presence-absence was calculated from data sheets (reflecting lab sorting) for surveys through 1990. During 1998-2004, L_p in the Choptank River was determined directly in the field and recorded on data sheets (P. Piavis, MD DNR, personal communication). All tows were made for two minutes. Standard 0.5 m diameter nets were used in the Nanticoke River during 1965-1971 (1.0 · 0.5 mm mesh) and after 1998 in the Choptank River (0.5 mm mesh). Trawls with 0.5 m nets (0.5 mm mesh) mounted in the cod-end were used in the Choptank River during 1980 -1990 (Uphoff 1997; Uphoff et al. 2005). Survey designs for the Choptank and Nanticoke Rivers were described in Uphoff (1997).

We used property tax map-based counts of structures per hectare (C/ha) in a watershed as our indicator of development (Uphoff et al. 2012). This indicator has been estimated for us by Marek Topolski of the Fishery Management Planning and Fish Passage Program. Tax maps are graphic representations of individual property boundaries and existing structures that help State tax assessors locate properties (Maryland Department of Planning or MDP 2010). All tax data were organized by county. Since watersheds straddle political boundaries, one statewide tax map was created for each year of available tax data, and then subdivided into watersheds. Maryland's tax maps are updated and maintained electronically as part of MDP's Geographic Information System's (GIS) database. Files were managed and geoprocessed in ArcGIS 9.3.1 from Environmental Systems Research Institute (ESRI 2009). All feature datasets, feature classes, and shapefiles were spatially referenced using the NAD_1983_StatePlane_Maryland_FIPS_1900 projection to ensure accurate feature overlays and data extraction. ArcGIS geoprocessing models were developed using Model Builder to automate assembly of statewide tax maps, query tax map data, and assemble summary data. Each year's statewide tax map was clipped using the MD 8-digit watershed boundary file, and modified to exclude estuarine waters, to create watershed land tax maps. These watershed tax maps were queried for all parcels having a structure built from 1700 to the tax data year. A large portion of parcels did not have any record of year built for structures but consistent undercounts should not present a problem since we are interested in the trend and not absolute magnitude (Uphoff et al. 2012).

Estimates of C/ha were used as a measure of watershed development intensity for analysis with L_p. Generally, whole watershed estimates were used with the following exceptions: Nanticoke and Choptank river watersheds were truncated at the lower boundaries of their striped bass spawning areas and at the Delaware border (latter due to lack of comparable data). Estimates of C/ha were available from 1950 through 2011 (M. Topolski, MDDNR, personal communication). Estimates of C/ha for 2011 were used to represent 2012 and 2013.

Estimates of C/ha for the IS target and limit were estimated from a power function that converts C/ha to IS based on Towson University satellite data interpretation (Uphoff et al. 2012). The target proposed in Uphoff et al. (2011a), 5.5% IS, was reduced to 5% to meet IS guideline being developed by Maryland's Department of Natural Resources (MDDNR 2012). The IS threshold of 10% in Uphoff et al. (2011a) remained unchanged. An estimate equivalent to 15% IS was also made to designate suburban watersheds that were developed well beyond the threshold. Estimates of C/ha that were equivalent to 5% IS, 10% IS, and 15% IS were estimated as 0.27, 0.83, and 1.59 C/ha, respectively by Uphoff et al. (2012).

Two regression approaches were used to examine the relationship between C/ha and L_p. First, separate linear regressions of C/ha against L_p were estimated for brackish and tidal-fresh subestuaries. If 95% CIs of slopes overlapped and 95% CIs of the intercepts did not overlap, we used the multiple regression of C/ha and salinity class against L_p. This latter approach assumed slopes were equal for two subestuary salinity categories, but intercepts were different (Freund and Littell 2006). Salinity was modeled as an indicator variable in the multiple regression with 0 indicating tidal-fresh conditions and 1 indicating brackish conditions. High salinity has been implicated in contributing to low L_p in Severn River (Uphoff et al. 2005). The association of mean salinity and IS can be significant and strong (Uphoff et al. 2010), and salinity is important to formation of stressful DO conditions in summer in mesohaline tributaries (see Section 3). Ricker (1975) warned against using well correlated variables in multiple regressions, so categorizing salinity for multiple or separate regressions of C/ha against L_p minimized confounding salinity with level of development. Level of significance was set at $\alpha \leq 0.05$. Residuals were inspected for trends, non-normality, and need for additional terms.

We used Akaike information criteria adjusted for small sample size, AIC_c, to evaluate the models that describe hypotheses that related changes in L_p to C/ha for each salinity category (separate slopes) or to C/ha and salinity category (common slopes, separate intercepts; Burnham and Anderson 2001):

$^{(4)}AIC_c = -2(\text{log-likelihood}) + 2K + [(2K \cdot (K+1)) / (n-K-1)]$;

where n is sample size and K is the number of model parameters. Model parameters for the least squares regressions consisted of their mean square error estimates (variance), intercepts, slopes, and salinity category in the case of the multiple regression. We rescaled AIC_c values to D_i, ($AIC_{c\,i}$ – minimum AIC_c), where i is an individual model, for the tidal-fresh or brackish regression compared to the multiple regression. The D_i values provided a quick "strength of evidence" comparison and ranking of models and hypotheses. Values of $D_i \leq 2$ have substantial support, while those > 10 have essentially no support (Burnham and Anderson 2001).

An additional view of the relationship of L_p and C/ha was developed by considering dominant land use classification when interpreting salinity classification (brackish or tidal-fresh), C/ha, and *Lp* regressions. Primary land use (forest, agriculture, or urban) was determined from Maryland Department of Planning estimates for 1973, 1994, 1997, 2002, or 2010 that fell closest to a sampling year. These latter categories were not used in regression analyses, but were considered in the interpretation of results. Urban land consisted of high and low density residential, commercial, and institutional acreages (MD DNR 1999).

The mean of feeding success rank was calculated annually for each subestuary sampled in 2010-2013, as was mean total length (TL in mm) of larvae. The proportion of guts without food (P0) was estimated for each subestuary as was the proportion of larvae with copepods (P_{cope}), cladocerans (P_{clad}), or other (P_{othr}) food items. The latter three proportions were not additive.

Associations of C/ha with mean feeding rank, P0, mean TL, P_{cope}, P_{clad}, and P_{othr} (2010-2013 estimates) were tested with correlation analysis. Correlations of L_p with P0 and mean feeding rank were used to evaluate whether larval relative abundance was associated with feeding success. An additional set of correlation analyses examined associations among mean feeding success rank, mean TL, P_{cope}, P_{clad}, and P_{othr}.

We used OM0 (proportion of samples without OM, i.e., rank = 0) as our indicator of detritus availability and correlated OM0 against C/ha and feeding parameters that were significantly associated with C/ha. Proportions of samples without OM were estimated during 2011-2013, so fewer observations were available for analysis. The distribution of OM ranks assigned to samples in 2011-2013 was highly skewed towards zero and few ranks greater than 1 were reported.

We were specifically interested in the relationships of the amount of organic matter to development and larval feeding success. Examination of the plot of OM0 and C/ha suggested that the relationship could be nonlinear, with OM0 increasing at a decreasing rate with C/ha. We fit a power and logistic growth functions to these data using Proc NLIN in SAS (Freund and Littel 2006). The power function was described by the equation:

$$^{(5)}OM0 = a \cdot (C/ha)^b;$$

where a is a scaling coefficient and b is a shape parameter. The logistic growth function was described by the equation:

$$^{(6)}OM0 = b / ((1 + ((b - c) / c) \cdot (\exp(-a \cdot C/ha)));$$

where a is the growth rate of OM0 with C/ha, b is maximum OM0, and c is OM0 at C/ha = 0 (Prager et al. 1989).

We used linear and quadratic regressions to explore relationships of feeding success (mean of feeding ranks) with OM0 (Freund and Littel 2006). Linear and quadratic regressions explored this relationship for all data, with the linear regression describing a hypothesis about steady change, while the dome-shaped quadratic relationship would indicate an optimum value of OM0 for feeding success. A linear regression was also used on points representing only forested and urban watersheds, removing larger, agricultural (the only watersheds dominated by agriculture) Eastern Shore watersheds from consideration and confined remaining comparisons to western shore subestuaries.

During 2013, we collected Yellow Perch larvae for RNA/DNA analysis from a regional urban gradient represented by the watersheds of Piscataway Creek (C/ha = 1.46), Mattawoman Creek (C/ha = 0.91), and Nanjemoy Creek (C/ha = 0.09; Figure 2-1). This design, based on several previous years' collections, anticipated that sampling from these three rivers on three occasions would provide 30 larvae per date for a total of 180.

Samples for RNA/DNA analysis were collected when larvae were gathered for analysis of gut contents. In the field, Yellow Perch larvae were composited from several stations (where possible) that bracketed where larvae are abundant. Once a candidate jar had been checked for larvae and OM, the sample was poured through a 500 µ screen and larvae were transferred to a large tube with special preservative (RNAlater®). The vial was labeled with the subestuary name and sample date. Larvae from other sites from one subestuary were composited into the vial on the same date.

In the lab, larvae for each date were processed for both gut contents and RNA/DNA ratios. Yellow Perch larvae 11 mm TL or less were examined for gut contents from each sample. These larvae represented first-feeding and early postlarvae, larvae that absorbed their yolk and began active feeding. Generally, 7 mm larvae were the smallest that contained food. Larvae were removed from the composite sample and placed in a Petri dish of water, examined for gut contents and then the guts were removed. The RNA/DNA ratio estimate did not contain food items. If a larva had not fed, the guts were teased away to be safe. Each processed larva was placed in a small individual vial of RNAlater preservative. The vial was coded on the outside as follows: letter designating which creek, number designating which sample date, and number designating which individual larva was placed in the vial.

RNA/DNA ratios were estimated by J. Brush at the Cooperative Oxford Laboratory. Protocols for estimating RNA/DNA generally followed Kaplan et al. (2001). Larvae were stored at 4°C in RNAlater® for a few weeks until ready for processing. Whole body samples, minus gut contents, were digested with 1% sodium dodecylsulfate, proteinase K digestion buffer (66ug/ml), and 0.1M NaCl at 55^{o}C for several hours until completely digested. Samples were centrifuged at 11,000 rpm for 10 minutes, and the supernatant containing the nucleic acids was removed and stored at -80^{o}C until ready for processing.

DNA was removed from a subsample of each sample's supernatant using 10X DNase digestion buffer (0.2M Tris-HCl pH=7.5, 0.1M MgCl and 0.02M CaCl) and RNase-free DNase I. Samples incubated at 37 ^{o}C for 45 minutes in a dry bath. Samples were centrifuged for 5 minutes at 8,000 rpm. The supernatant was removed and stored at -80 °C until ready for processing.

Samples were fluorometrically analyzed for DNA and RNA quantification using a 96-well microplate, 45% TE Buffer, 50% Quant-it™ PicoGreen® for DNA and 50% Quant-it™ RiboGreen® for RNA (Molecular Probes, Oregon), and Synergy 2 microplate reader. Samples were analyzed in triplicate using a black microplate (Corning). Fluorescence was measured at 480 nm excitation and 520 nm emission for both DNA and RNA quantification.

The RNA/DNA ratios for each subestuary were plotted against larval TL or date. Reference lines indicating starving (RNA/DNA < 2; Blom et al. 1997) and fed larvae (RNA/DNA > 3; Buckley 1984; Wright and Martin 1985) based on values from larvae of several marine species and Striped Bass were added to the plots. A tabular summary of C/ha, median RNA/DNA ratio, mean fullness rank, N, N < 2, and N > 3 was constructed. The proportions of larvae with RNA/DNA ratios less than 2 (proportion starved or *Ps*) and their 90% confidence intervals were estimated for each subestuary as

$$^{(7)}Ps = N_{<2} / N_{total};$$

where $N_{<2}$ equaled the number of samples with RNA/DNA ratios less than 2 and N_{total} equaled the total number of RNA/DNA samples. The SD of *Ps* was estimated as ;

$$^{(8)}SD = [(Ps\ (1- Ps)) / N_{total}]^{0.5} \text{ (Ott 1977).}$$

The 90% confidence intervals were constructed as

$$^{(9)}Ps \pm (1.44\ SD; \text{Ott 1977}).$$

Proportions of larvae with RNA/DNA ratios greater than 3 (proportion fed or *Pf*) were estimated as in equations 6-8, but *Pf* was estimated with the number of larvae with RNA/DNA ratios greater than 3 ($N_{>3}$) in the numerator of equation 6 and *Pf* was substituted for *Ps* in the remaining equations.

For each subestuary and sample date, RNA/DNA ratio means and the number of samples in the *Pf* and *Ps* categories were summarized along with mean fullness rank, mean TL, and total sample size. *Ps* and *Pf* and their 90% CI's were estimated (Johnson 1999). Confidence interval comparisons were limited to larvae with a common TL range among all subestuaries.

Results

During 2013, sampling began on March 27 in Piscataway, Mattawoman and Nanjemoy creeks, and they were sampled through April 25; samples through April 22 were used to estimate L_p. Sampling began on March 19 in the Northeast River and ended on April 23. It should be noted that sampling on the Northeast River ended before a decline in presence was seen (Yellow Perch larvae were present at all 10 stations on

April 23). The estimate for this system may be biased high in 2013. In past years, L_p on the Northeast River was estimated from samples collected through April 29, 2010, April 22, 2011, and April 12, 2012. These dates represented a drop in larval presence, therefore, a cut-off of April 23 was compatible with two of three estimates of L_p made for Northeast River. Nanticoke River was sampled between April 3 and 30 and samples taken during April 8-26 were used to estimate L_p. Bush River was first sampled on March 19 and last sampled on April 25; dates between April 4 and 25 were used for estimating L_p. Choptank River was first sampled on March 28 and last sampled May 16; dates between April 2 and April 26 were used for estimating L_p. Patuxent River was sampled on seven visits between March 20 and April 26. Sampling in the Patuxent River was inconsistent (8 sites per date) and was

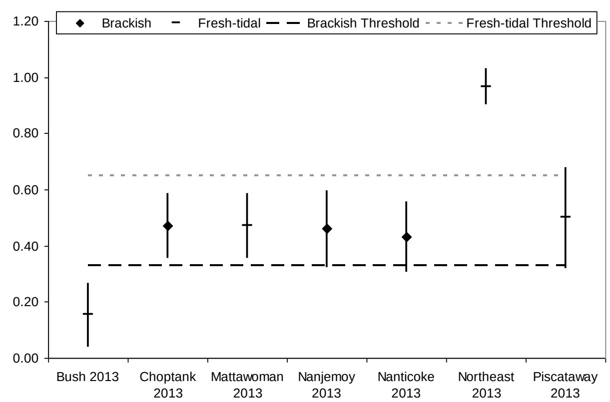

Figure 2-2. Proportion of tows with larval Yellow Perch (*L*p) and its 95% confidence interval in systems studied during 2013. Mean *L*p of brackish tributaries indicated by diamond and tidal-freah mean indicated by dash.

not used to estimate L_p.

Based on 95% CIs, estimates of L_p during 2013 were judged sufficiently precise to detect significant differences among subestuaries (Figure 2-2). Estimates of L_p for brackish subestuaries (Nanjemoy Creek, Nanticoke River, and Choptank River) were similar to estimates for tidal-fresh subestuaries (Mattawoman and Piscataway Creeks) in 2013 (range 0.43 to 0.50). During 2013, there were low risks (0-3.5%) that Nanjemoy Creek, Northeast River (with reservations about positive bias), and Nanticoke River fell below their L_p thresholds. These were subestuaries with rural watersheds. The three subestuaries with suburban watersheds all exhibited high risks of falling below their thresholds (93.5% - 100%).

None of the brackish subestuaries sampled during 2013 fell persistently below the threshold L_p (Figure 2-

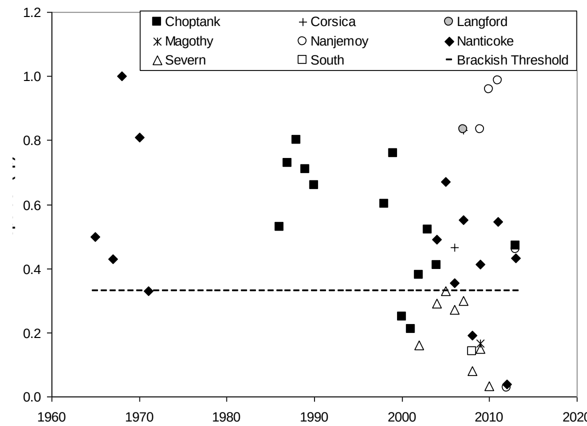

Figure 2-3. Proportion of tows with Yellow Perch larvae (*Lp*) for brackish subestuaries, during 1965-2013. Dotted line provides reference for persistent poor *Lp* exhibited in developed brackish subestuaries.

3). Tidal-fresh Bush River, Mattawoman Creek, and Piscataway Creek (C/ha > 0.91) have exhibited low L_p for two years in a row, while Northeast River (C/ha = 0.46) has not (Figure 2-4).

The range of C/ha values available for analysis with L_p was more shifted towards the y-axis in brackish subestuaries (C/ha range = 0.05-2.73) than tidal-fresh (0.45-3.33; Table 2-1). None of the tidal-fresh estuaries analyzed were at or below the target condition (Table 2-1).

Separate linear regressions of C/ha and L_p by salinity category were significant at $P \leq 0.0006$; Table 2-2).

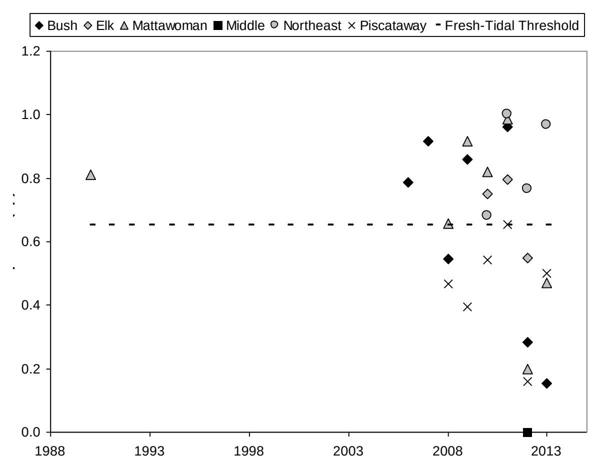

Figure 2-4. Proportion of tows with Yellow Perch larvae (*Lp*) for tidal-fresh subestuaries, during 1990-2013. Dotted line provides reference for consistent poor *Lp* exhibited in a more developed tidal-fresh subestuary (Piscataway Creek).

River	Year	C / ha	Primary Land Use	Salinity	Lp
Bush	2006	1.17	Urban	0	0.79
Bush	2007	1.19	Urban	0	0.92
Bush	2008	1.2	Urban	0	0.55
Bush	2009	1.21	Urban	0	0.86
Bush	2011	1.22	Urban	0	0.96
Bush	2012	1.23	Urban	0	0.28
Bush	2013	1.23	Urban	0	0.15
Choptank	1986	0.09	Agriculture	1	0.53
Choptank	1987	0.09	Agriculture	1	0.73
Choptank	1988	0.1	Agriculture	1	0.8
Choptank	1989	0.1	Agriculture	1	0.71
Choptank	1990	0.1	Agriculture	1	0.66
Choptank	1998	0.13	Agriculture	1	0.6
Choptank	1999	0.13	Agriculture	1	0.76
Choptank	2000	0.13	Agriculture	1	0.25
Choptank	2001	0.13	Agriculture	1	0.21
Choptank	2002	0.14	Agriculture	1	0.38
Choptank	2003	0.14	Agriculture	1	0.52
Choptank	2004	0.15	Agriculture	1	0.41
Choptank	2013	0.16	Agriculture	1	0.47
Corsica	2006	0.21	Agriculture	1	0.47
Corsica	2007	0.22	Agriculture	1	0.83
Elk	2010	0.59	Forest	0	0.75
Elk	2011	0.59	Forest	0	0.79
Elk	2012	0.59	Forest	0	0.55
Langford	2007	0.07	Agriculture	1	0.83
Magothy	2009	2.73	Urban	1	0.17
Mattawoman	1990	0.45	Forest	0	0.81
Mattawoman	2008	0.87	Forest	0	0.66
Mattawoman	2009	0.88	Forest	0	0.92
Mattawoman	2010	0.9	Forest	0	0.82
Mattawoman	2011	0.91	Forest	0	0.99
Mattawoman	2012	0.91	Forest	0	0.2
Mattawoman	2013	0.91	Forest	0	0.47
Middle	2012	3.33	Urban	0	0
Nanjemoy	2009	0.09	Forest	1	0.83
Nanjemoy	2010	0.09	Forest	1	0.96
Nanjemoy	2011	0.09	Forest	1	0.99
Nanjemoy	2012	0.09	Forest	1	0.03
Nanjemoy	2013	0.09	Forest	1	0.46
Nanticoke	1965	0.05	Agriculture	1	0.5
Nanticoke	1967	0.05	Agriculture	1	0.43
Nanticoke	1968	0.05	Agriculture	1	1
Nanticoke	1970	0.06	Agriculture	1	0.81
Nanticoke	1971	0.06	Agriculture	1	0.33
Nanticoke	2004	0.11	Agriculture	1	0.49
Nanticoke	2005	0.11	Agriculture	1	0.67
Nanticoke	2006	0.11	Agriculture	1	0.35
Nanticoke	2007	0.11	Agriculture	1	0.55
Nanticoke	2008	0.11	Agriculture	1	0.19
Nanticoke	2009	0.11	Agriculture	1	0.41
Nanticoke	2011	0.11	Agriculture	1	0.55
Nanticoke	2012	0.11	Agriculture	1	0.04
Nanticoke	2013	0.11	Agriculture	1	0.43
Northeast	2010	0.46	Forest	0	0.68
Northeast	2011	0.46	Forest	0	1
Northeast	2012	0.46	Forest	0	0.77
Northeast	2013	0.46	Forest	0	0.97
Piscataway	2008	1.41	Urban	0	0.47
Piscataway	2009	1.43	Urban	0	0.39
Piscataway	2010	1.45	Urban	0	0.54
Piscataway	2011	1.46	Urban	0	0.65
Piscataway	2012	1.46	Urban	0	0.16
Piscataway	2013	1.46	Urban	0	0.5
Severn	2002	2.02	Urban	1	0.16
Severn	2004	2.09	Urban	1	0.29
Severn	2005	2.15	Urban	1	0.33
Severn	2006	2.18	Urban	1	0.27
Severn	2007	2.21	Urban	1	0.3
Severn	2008	2.24	Urban	1	0.08
Severn	2009	2.25	Urban	1	0.15
Severn	2010	2.26	Urban	1	0.03
South	2008	1.61	Urban	1	0.14

Table 2-1. Estimates of proportions of ichthyoplankton net tows with Yellow Perch larvae (L_p) during 1965-2013 and data used for regression with counts of structures per hectare (C/ha). Salinity class 0=tidal-fresh ($\leq$ 2.0 ‰) and 1=brackish (> 2.0 ‰). Year is the year a subestuary was sampled. Primary land use was determined from Maryland Department of Planning estimates for 1973, 1994, 1997, 2002, or 2010 that were closest to a sampling year. These latter categories were not used in regression analyses.

ANOVA			Brackish			
Source	df	SS	MS	F	P	
Model	1	0.98558	0.98558	19.01	<.0001	
Error	43	2.2299	0.05186			
Total	44	3.21548				
r^2	0.3065					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.56585	0.04056	13.95	<.0001	0.48405	0.64764
C / ha	-0.17049	0.03911	-4.36	<.0001	-0.24936	-0.09162

ANOVA			Tidal-Fresh			
Source	df	SS	MS	F	P	
Model	1	0.79071	0.79071	15.05	0.0006	
Error	26	1.36584	0.05253			
Total	27	2.15655				
r^2	0.3667					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.94771	0.09302	10.19	<.0001	0.7565	1.13891
C / ha	-0.29918	0.07712	-3.88	0.0006	-0.4577	-0.14067

ANOVA			Multiple Regression			
Source	df	SS	MS	F	P	
Model	2	2.09792	1.04896	19.78	<.0001	
Error	70	3.71181	0.05303			
Total	72	5.80973				
r^2	0.3611					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.83872	0.05751	14.58	<.0001	0.72402	0.95342
C / ha	-0.19709	0.03522	-5.6	<.0001	-0.26734	-0.12684
Salinity	-0.25778	0.05816	-4.43	<.0001	-0.37377	-0.14179

Table 2-2. Summary of results of regressions of proportions of tows with Yellow Perch larvae (L_p) and counts of structures per hectare (C/ha). Separate regressions by salinity (tidal-fresh ≤ 2.0 ‰ and brackish > 2.0 ‰) and a multiple regression using salinity as a class variable (tidal-fresh = 0 and brackish = 1) are presented.

These analyses indicated that C/ha was negatively related to L_p and L_p was, on average, higher in tidal-fresh subestuaries than in brackish subestuaries.

Estimates of C/ha accounted for 31% of variation of L_p in brackish subestuaries and 37% in tidal-fresh subestuaries. Based on 95% CI overlap, intercepts were significantly different between tidal-fresh (mean = 0.94, SE = 0.09) and brackish (mean = 0.57, SE = 0.04) subestuaries. Mean slope for C/ha estimated for tidal-fresh subestuaries (mean = -0.30, SE = 0.08) were steeper, but 95% CI's overlapped CI's estimated for the slope of brackish subestuaries (mean = -0.17, SE = 0.04; Table 2-2). Both regressions indicated that L_p would be extinguished between 3.0 and 3.5 C/ha (Figure 2-5).

Overall, the multiple regression approach offered a similar fit (R^2 = 0.36; Table 2-2) to separate regressions for each type of subestuary. Intercepts of tidal-fresh and brackish subestuaries equaled 0.95 and 0.57, respectively; the common slope was -0.20. Predicted L_p over the observed ranges of C/ha would decline from 0.56 to 0.10 in brackish subestuaries and from 0.81 to -0.05 in tidal-fresh subestuaries (Figure 2-5).

Akaike's Information Criteria values equaled 9.5 for the regression of C/ha and L_p for brackish subestuaries, 9.9 for tidal-fresh estuaries, and 11.5 for the multiple regression that included salinity category. Calculations of D_i for brackish or tidal-fresh versus multiple regressions were approximately 1.98 and 1.58 (respectively), indicating that either hypothesis (different intercepts for tidal-fresh and brackish subestuaries with different or common slopes describing the decline of L_p with C/ha) were plausible.

Although we have analyzed these data in terms of tidal-fresh and brackish subestuaries, inspection of Table 2-1 indicated an alternative interpretation based on primary land use estimated by MDP. Predominant

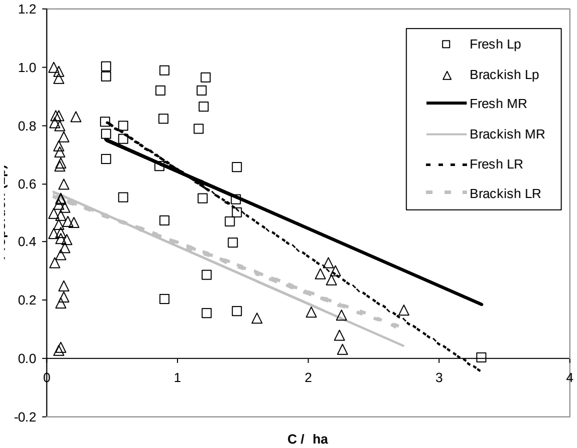

Figure 2-5. Relationship of proportion of plankton tows with Yellow Perch larvae and development (structures per hectare or C/ha) indicated by multiple regression of tidal -fresh and brackish subestuaries combined (prediction = MR) and separate linear regressions for both (prediction = LR).

land use at lowest levels of development may be influencing the intercept estimates. Rural watersheds were absent for tidal-fresh subestuaries analyzed and the lowest levels of development were dominated by forest (Figure 2-6). Nearly all rural land in brackish tributaries was dominated by agriculture. Dominant land cover estimated by MD DOP for watersheds of tidal-fresh subestuaries was equally split between forest (C/ha = 0.46-0.91) and urban (C/ha $\geq$ 1.17; 14 observations each). Brackish subestuary watershed rural lands were dominated by agriculture (C/ha $\leq$ 0.22; 30 observations), while forest land cover (C/ha ~ 0.09) was represented by five observations. The range of L_p was similar in brackish subestuaries with forest and agricultural cover, but the distribution seemed shifted towards higher L_p in the limited sample from the forested watershed (Nanjemoy Creek). Urban land cover predominated in nine watersheds of brackish subestuaries (C/ha $\geq$ 1.61; Table 2-1; Figure 2-6). Tidal-fresh subestuary intercepts may have represented the intercept for forest cover and brackish subestuary intercepts may have represented agricultural influence.

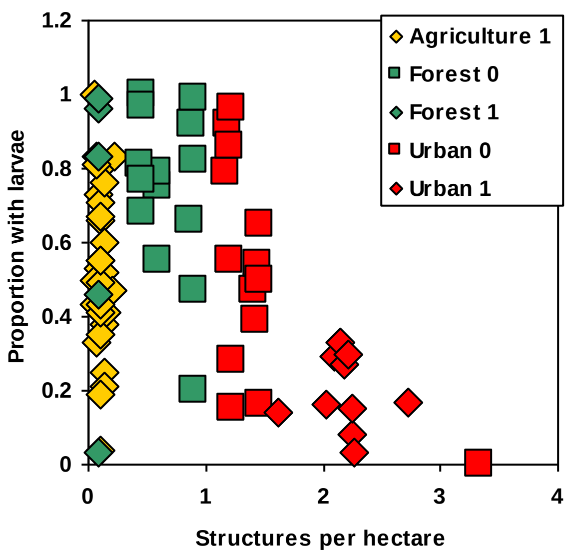

Figure 2-6. Proportion of plankton tows with Yellow Perch larvae plotted against development (C/ha) with Department of Planning land use designations and salinity class indicated by symbols. Diamonds indicate brackish subestuary and squares indicate tidal-fresh. A "1" behind land use in the key indicated brackish, and a "0" tidal-fresh.

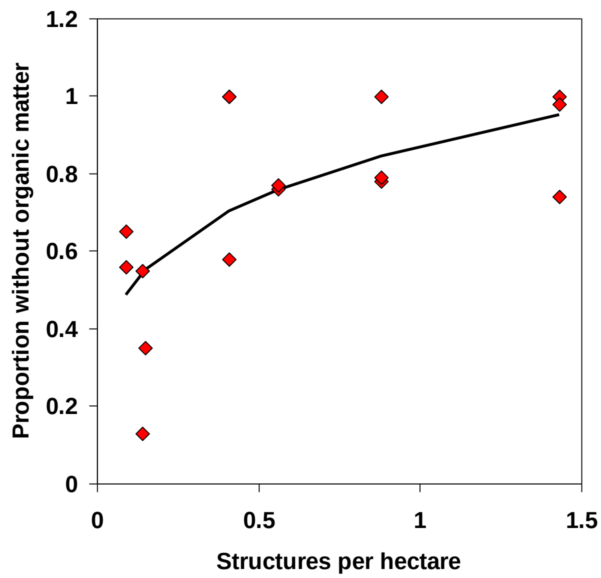

Figure 2-7. Relationship of proportion of plankton tows without detritus (OM0) and development (structures per hectare or C / ha).

If this is the case, then forest cover provides for higher L_p than agriculture. Increasing suburban land cover leads to a significant decline in L_p regardless of rural land cover type.

We examined 332 larval guts during 2010, 523 in 2011, 466 in 2012, and 706 in 2013. Samples were drawn primarily from tidal-fresh subestuaries (16 of 22 subestuary and year combinations). A smaller sample size was available for correlations with OM0 (N = 16) than other variables (N = 22) because observations of OM did not start until 2011.

Larvae averaged 8.11.1 mm in 2010 (size range among days sampled), 8.3-9.3 mm in 2011, 7.5-8.8

River	Year	C / ha	Mean full	OM0	P0	Pclad	Pcope	Pother	Mean TL	N
Elk	2010	0.59	2.75		0.05	0.02	0.95	0.13	11.1	110
Mattawoman	2010	0.9	2		0.09	0.15	0.78	0.09	9.2	55
Nanjemoy	2010	0.09	2.88		0	0.1	1	0.15	9.1	48
Northeast	2010	0.46	2.34		0.19	0.22	0.72	0.3	8.4	64
Piscataway	2010	1.45	1.85		0.13	0	0.55	0.53	9.4	55
Elk	2011	0.59	2.81	0.76	0.08	0	0.96	0.01	8.9	90
Mattawoman	2011	0.91	0.9	0.78	0.42	0.02	0.51	0.07	9.3	110
Nanjemoy	2011	0.09	2.18	0.56	0.07	0.03	0.83	0.2	9	150
Nanticoke	2011	0.11	3.27	0.55	0.08	0.71	0.92	0.16	8.6	51
Northeast	2011	0.46	2.44	0.58	0.08	0	0.91	0.09	8.3	90
Piscataway	2011	1.46	0	1	1	0	0	0	8.4	32
Bush	2012	1.23	2.48		0	0.55	0.53	1	8.6	40
Elk	2012	0.59	0.77	0.77	0.24	0.02	0	0.7	7.7	198
Mattawoman	2012	0.91	1.81	1	0	0.44	0.88	1	8.8	16
Northeast	2012	0.46	1.17	0.99	0.01	0.04	0.08	0.99	7.5	203
Piscataway	2012	1.46	1.67	0.98	0	0.56	0.67	1	8.7	9
Choptank	2013	0.16	1.04	0.33	0.21	0.37	0.34	0.33	7.6	319
Mattawoman	2013	0.91	1.69	0.79	0	0.84	0.69	0.04	7.6	98
Nanjemoy	2013	0.09	1.59	0.65	0	0.59	0.42	0.23	7.3	64
Nanticoke	2013	0.11	1.08	0.13	0.33	0.4	0.25	0.23	8.3	132
Northeast	2013	0.46	0.55	1	0.46	0.2	0	0.44	8.8	80
Piscataway	2013	1.46	2.31	0.74	0	0.38	0.69	0.23	7.9	13

Table 2-3. Summary of estimates used in correlation analysis of Yellow Perch larval feeding success. C / ha = counts of structures per hectare. Mean full = mean of fullness ranks assigned to larval guts. OM0 = proportion of samples without organic matter (detritus). P0 = proportion of guts without food. Pclad = proportion of guts with cladocerans. Pcope = proportion of guts with copepods. Pother = proportion of guts with "other" food items. Mean TL = mean TL of larvae in mm. Mean fullness = average feeding rank of larvae. N = number of Yellow Perch larvae examined.

mm in 2012, and 7.3-8.8 mm in 2013 (Table 2-3). Larval Yellow Perch guts contained food in all years and subestuaries except Piscataway Creek during 2011. Copepods were the most prevalent food item during 2010 and 2011, and were found in 51-100% of guts sampled (excluding Piscataway Creek). Copepods were not as prevalent in 2012 and only Piscataway and Mattawoman Creeks had P_{cope} estimates within the range observed in 2010-2011. In 2013, copepods were still not as prevalent and were found in 0-69% of guts sampled (Northeast River larvae did not contain copepods). Cladocerans were found in a higher proportion of guts sampled in 2013 (20-84%) then in guts during 2010-2012 (0-56%), with the exception of the Nanticoke River in 2011 (71%). The "other" food item category represented a high fraction of guts in Piscataway Creek (53%) in 2010 and 1-30% of guts in remaining subestuaries during 2010-2011. This category was predominant in larval gut samples from all five subestuaries during 2012, but it should be noted that most gut contents were already too digested to be identifiable and could not be categorized any other way during that year (70-100%; Table 2-3). In 2013 gut content identification was more straightforward and the "other" food item category represented what was seen in previous years (44%).

During 2010-2013, percentage of guts without food ranged from 0 to 46% in all subestuary and year combinations except Piscataway Creek during 2011 (100%). Mean fullness rank ranged between 0.6 and 3.3 in all subestuary and year combinations except Piscataway Creek during 2011 (where it was 0; Table 2-3). In comparison with 2010 and 2011, feeding success was low in both 2012 and 2013 (Table 2-3).

The type of food present in larval Yellow Perch guts was significantly associated OM, but not with development. P_{cope} was negatively correlated with OM0 (r = -0.54, P = 0.05), while P_{other} was positively correlated (r = 0.57, P = 0.03; Table 2-4). The amount of food present in larval guts was also correlated with the presence of copepods, with both mean fullness rank and P0 being significantly associated with P_{cope} (r = 0.88, P = <0.0001 and r = -0.60, P = 0.006, respectively). Mean TL was positively correlated with P_{cope} (r = 0.44, P = 0.05), indicating larger larvae had copepods present in their diets more often. Estimates of L_p were significantly and negatively correlated with P_{clad} (r = -0.68, P = 0.001) and P_{othr} (r = -0.60, P = 0.005; Table 2-4).

Estimates of C/ha and OM0 were significantly related. A non-linear power function fit the data (approximate r^2 = 0.46, α < 0.0001; N = 16), depicting OM0 increasing towards 1.0 at a decreasing rate as C/

Parameter	Statistic	C / ha	Mean Fullness	OM0	P0	P_{clad}	P_{cope}	P_{other}	Mean TL
	r	-0.31							
	P	0.18							
Mean Fullness	N	20							
	r	0.57	-0.66						
	P	0.03	0.01						
OM0	N	20	14						
	r	0.25	-0.7	0.36					
	P	0.29	0.0005	0.2					
P0	N	20	20	14					
	r	0.07	0.21	-0.08	-0.38				
	P	0.78	0.37	0.78	0.1				
P_{clad}	N	20	20	14	20				
	r	-0.2	0.88	-0.54	-0.6	0.16			
	P	0.4	<.0001	0.05	0.006	0.5			
P_{cope}	N	20	20	14	20	20			
	r	0.26	-0.17	0.57	-0.29	0.2	-0.3		
	P	0.26	0.48	0.03	0.22	0.41	0.2		
P_{other}	N	20	20	14	20	20	20		
	r	0.06	0.31	0.04	0.04	-0.39	0.44	-0.19	
	P	0.81	0.18	0.9	0.88	0.09	0.05	0.42	
Mean TL	N	20	20	14	20	20	20	20	
	r	-0.5	-0.02	-0.31	0.27	-0.68	0.03	-0.6	0.27
	P	0.02	0.95	0.29	0.26	0.001	0.91	0.005	0.25
L_p	N	20	20	14	20	20	20	20	20

Table 2-4. Correlation matrix for Yellow Perch larval feeding success. C/ha = counts of structures per hectare. Mean fullness = average feeding rank of larvae. OM0 = proportion of samples without organic matter. P0 = proportion of guts without food. Pclad = proportion of guts with cladocerans. Pcope = proportion of guts with copepods. Pothr = proportion of guts with “other” food items. Mean TL = mean TL of larvae in mm. L_p = proportion of plankton tows with larvae. Statistic r = Pearson correlation coefficient, P = level of significance, and N = number of observations. Gray shading indicates correlation of interest at P $\leq$ 0.05.

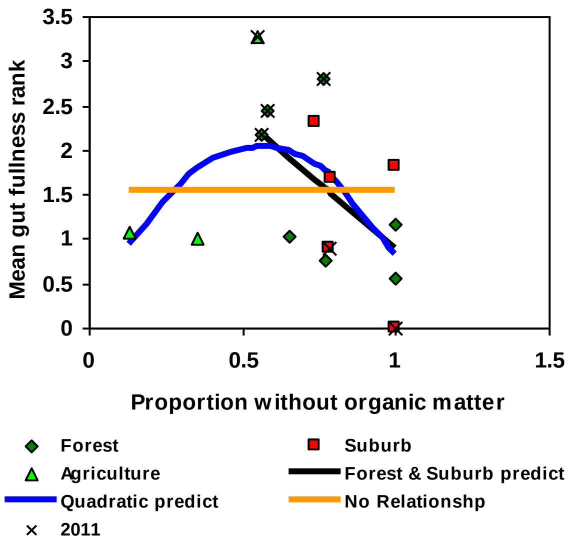

Figure 2-8. Suggested relationship of mean fullness rank of larval Yellow Perch and proportion of plankton tows without detritus (OM0) during 2011 -2013. Symbols with an "X" indicate values for 2011.

ha approached 1.50 (Figure 2-8). The relationship was depicted by the equation (5).

Approximate standard errors were 0.06 and 0.08 for parameters *a* and *b*, respectively. The logistic growth function (equation 6) fit these data similarly, but term *a* was not significantly different from zero.

Regression analyses indicated that organic matter may have a limited influence on larval feeding success, at best. A linear regression of OM0 and mean fullness rank using all data (agricultural, forest, and urban watersheds) was not significant (r^2 = 0.05, α = 0.41, N = 15; Figure 2-8) and did not indicate that OM0 influenced feeding success of Yellow Perch larvae. A linear regression of subset of watersheds (western shore subestuaries that were forested or urban, omitting Eastern Shore agricultural watersheds) explained about 30% of variation in feeding success (α = 0.06, N = 12; Figure 2-8). A dome-shaped quadratic regression to all data fit about as well (r^2 = 0.32, α = 0.10, N = 15; Figure 2-8) as the forest and urban subset fit to the linear regression. The descending portion of the quadratic model was consistent with the decline of forest and urban subset described above. The quadratic model suggested an optimum level of OM0 of about 0.55 that would produce a predicted mean fullnessof 2.05 (Figure 2-8). Curve fitting OM0 and feeding success data may not have provided a means of understanding a phenomenon (high feeding success) that occurs episodically when first-feeding Yellow Perch larvae and abundant copepods match. We believe 2011 represents a year where timing of Yellow Perch larvae and copepods matched, enhancing feeding success over a broad geographic area (Nanticoke River, Nanjemoy Creek, Elk River and Northeast River encompassed the lower Eastern Shore, Potomac River, and upper Chesapeake Bay). In 2011, four of five mean fullness ranks greater than 2 were encountered during the span that OM was measured (2011-2013). The two suburban watersheds sampled in 2011, both tributaries of the Potomac River, clearly had higher OM0 and low feeding scores, while rural watersheds had high mean fullness ranks and lower OM0 (Figure 2-8). Remaining years with mostly lower feeding success did not exhibit a clear pattern of feeding success with OM0 and it was likely that timing of zooplankton did not match first-feeding larvae. Anecdotally, we observed that high copepod abundance in sample jars peaked earlier than Yellow Perch larvae during 2013.

Yellow Perch larvae were collected (as designed) for RNA/DNA analysis from Mattawoman (N = 97: C/ha = 0.91), Piscataway (N = 10: C/ha = 1.46), and Nanjemoy creeks (N = 63; C/ha = 0.09) on April 8, 10, and 15, 2013. Mattawoman Creek collections had larvae as large as 10.5 mm. Collections from Nanjemoy and Piscataway creeks did not have larvae greater than 9.5 mm (Figure 2-9). Nanjemoy Creek's watershed was below the threshold development level, while the two remaining watersheds had passed the suburban threshold. Estimates of OM0 were 0.65 in Nanjemoy Creek, 0.79 in Mattawoman Creek, and 0.74 in Piscataway Creek.

Ratios of RNA/DNA were highest for 5.5-7 mm TL postlarvae during 2013, but a ratio greater than 3 (well fed larvae based on marine and Striped Bass larvae) was found for a single, 7 mm larva (Figure 2-9). A single larva larger than 7.5 mm had an RNA/DNA ratio not indicative of starvation (ratio = 2.5). Ratios of RNA/DNA declined with TL in Mattawoman Creek to 0.5 or less for larvae 9 mm or larger (Table 2-5; Figure 2-9). Ratios increased with TL in Nanjemoy and at 8 mm few were as low as 0.5, but only one was greater than 2.0. Larvae were only encountered in Piscataway Creek on April 15, 2013, and ratios were within a similar range as those exhibited in Nanjemoy Creek on the same date (Table 2-5; Figure 2-9).

Estimates of *Pf* were zero in Nanjemoy and Piscataway creeks, and 0.02 in Mattawoman Creek. Too few larvae in the fed category were collected to estimate CI's. Estimates of *Ps* were 0.90 (90% CI = 0.85-95; C/ha = 0.91) in Mattawoman Creek in 2013, 0.98 in Nanjemoy Creek (90% CI = 0.95-1.00; C/ha = 0.09), and 0.89 in Piscataway Creek (90% CI = 0.73-1.00; C/ha = 1.43). Construction of 90% CI's was confined to

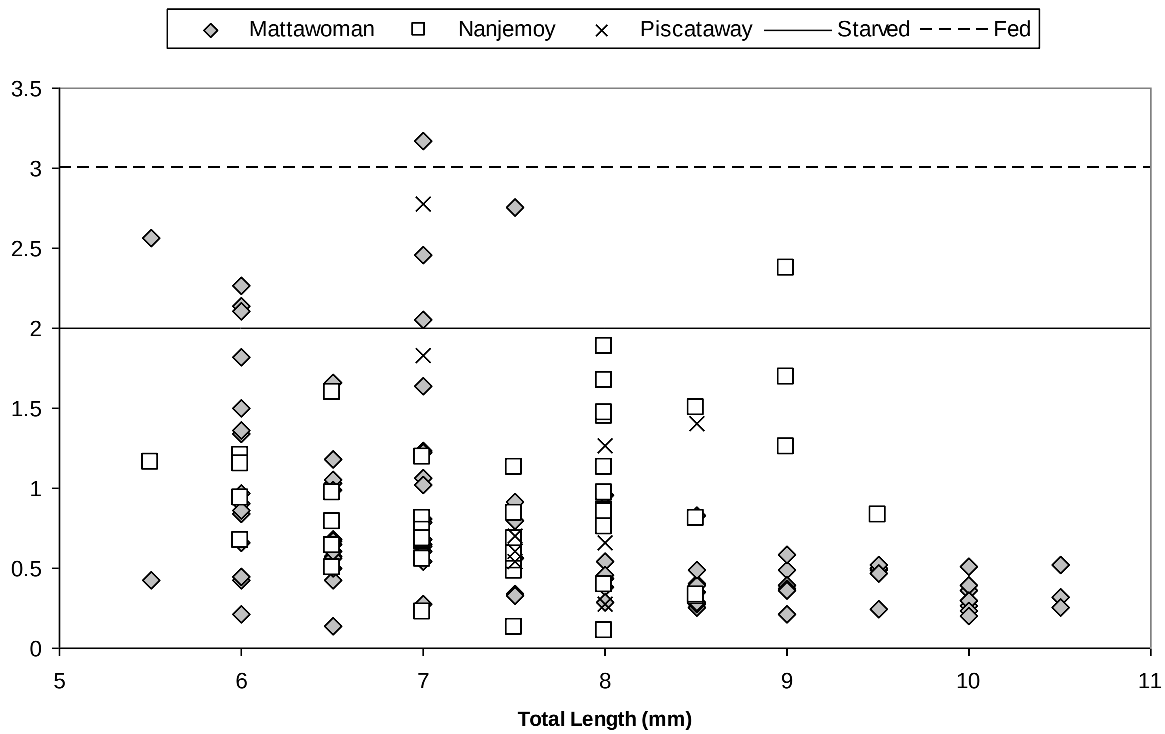

Figure 2-9. RNA/DNA ratios for Yellow Perch larvae by total length. Larvae were collected during 2013. Subestuaries are indicated by symbols. Reference lines are provided for ratios indicative of starved and fed conditions.

Ps of 6-9 mm TL larvae, the size in common in all three systems. The great majority of larvae collected during 2013 would have been considered in starved condition under the criterion developed from other fish larvae regardless of level of development or OM0.

Discussion

Estimates of L_p declined perceptibly once development exceeded the threshold (0.83 C/ha or 10% IS). A forest cover classification in a watershed was associated with higher L_p (median $L_p = 0.79$) than agriculture (median $L_p = 0.51$) or development (median $L_p = 0.30$). Interpretation of the influence of salinity class or primary land cover on L_p needs to consider that our survey design was limited to existing patterns of development. All estimates of L_p at or below target levels of development (0.27 C/ha or 5% IS; forested and agricultural watersheds) or at and beyond high levels of development (1.59 C/ha or 15% IS; urban watersheds) were from brackish subestuaries; estimates of L_p for development between these levels were from tidal-fresh subestuaries with forested watersheds. Larval dynamics below the target level of development primarily reflected Eastern Shore agricultural watersheds. Two types of land use would be needed to balance analyses: (1) primarily agricultural, tidal-fresh watersheds with below target development and (2) forested, brackish watersheds with development between the target and threshold. We do not believe that these combinations exist where Yellow Perch spawning occurs in Maryland's portion of Chesapeake Bay.

Salinity may restrict L_p in brackish subestuaries by limiting the amount of available low salinity habitat over that in a tidal-fresh subestuaries. Uphoff (1991) found that 90% of larvae collected in Choptank River during 1980-1985 were from 1‰ or less. Mortality of Yellow Perch eggs and prolarvae in experiments generally increased with salinity and was complete by 12‰ (Sanderson 1950; Victoria et al. 1992). The range of suitable salinities for prolarvae was lower than that for eggs (Victoria et al. 1992).

Development was an important influence on Yellow Perch egg and larval dynamics and negative changes generally conformed to ISRP guidelines in Uphoff et al. (2011a). Hilborn and Stokes (2010) advocated setting reference points related to harvest for fisheries (stressor) based on historical stock performance (outcome) because they were based on experience, easily understood, and not based on modeling. We believe applying IS or C/ha watershed development

Subestuary	Variable	8-Apr	10-Apr	15-Apr
	Mean Fullness	1.1	1.9	1.6
	Mean TL	6.2	6.8	9.1
	N	13	47	37
	Mean RNA/DNA	1	1.08	0.39
	SE RNA/DNA	0.19	0.11	0.02
	N RNA/DNA > 3	0	1	0
Mattawoman	N RNA/DNA < 2	12	34	36
	Mean Fullness	1.4	1	2
	Mean TL	6.5	6.8	8
	N	17	15	31
	Mean RNA/DNA	0.89	0.66	0.98
	SE RNA/DNA	0.08	0.11	0.09
	N RNA/DNA > 3	0	0	0
Nanjemoy	N RNA/DNA < 2	10	10	27
	Mean Fullness			2.4
	Mean TL			7.7
	N			10
	Mean RNA/DNA			1.12
	SE RNA/DNA			0.27
	N RNA/DNA > 3			0
Piscataway	N RNA/DNA < 2			8

Table 2-5. Summary of feeding success, larval length, sample size, and RNA/DNA characteristics, by subestuary and sample date. Data for dates with feeding information only and with RNA/DNA analysis are summarized. Mean fullness = mean feeding rank. Mean TL is in mm. N = total sample size of larvae processed for gut contents. Mean RNA/DNA is the average for the date. SE RNA/DNA is the standard error for the date. N RNA/DNA > 3 is the number of ratios above the fed criterion. N RNA/DNA < 2 is the number of ratios below the starvation criterion.

reference points (stressor) based on L_p (outcome) conforms to the approach advocated by Hilborn and Stokes (2010).

There appears to be some potential for development to influence organic matter (OM) and larval Yellow Perch feeding dynamics. However, OM may not matter much if there is not a match in the timing of copepod abundance and early feeding stages of Yellow Perch larvae. Timing of larvae and zooplankton abundance was an important aspect for the formation of strong year-classes of Striped Bass and White Perch (Limburg et al. 1999; Martino and Houde 2010). This analysis suggests that an influence of OM delivery on larval feeding success in urban watersheds may be episodic, occurring during years of high OM transport coupled with favorable timing of zooplankton and Yellow Perch larvae in rural watersheds.

Uphoff et al. (2013) found March temperature conditions also influenced L_p, and multiple regression models provided evidence that widespread climate factors (March precipitation as a proxy for OM transport and air temperature) influenced survival of Yellow Perch egg and larvae in Chesapeake Bay subestuaries and also supported the OM hypothesis. Yellow Perch require a period of low temperature for reproductive success (Heidinger and Kayes 1986; Ciereszko et al. 1997) and warm temperatures may preclude that from occurring.

Years of high spring discharge favor anadromous fish recruitment in Chesapeake Bay (Hoffman et al. 2007; Martino and Houde 2010) and may represent episodes of hydrologic transport of accumulated OM from watersheds (McClain et al. 2003) that fuel zooplankton production and feeding success. Under natural conditions, riparian marshes and forests would provide OM subsidies in high discharge years (Hoffman et al. 2007), while phytoplankton would be the primary source of OM in years of lesser flow. Stable isotope signatures of York River, Virginia, American shad larvae and zooplankton indicated that terrestrial OM largely supported one of its most successful year-classes. Lesser year-classes of American shad on the York River were associated with low flows, OM largely based on phytoplankton, and lesser zooplankton production (Hoffman et al. 2007). The York River watershed, with large riparian marshes and forest, was largely intact relative to other Chesapeake Bay tributaries (Hoffman et al. 2007).

Zooplankton supply (cladocerans and copepods) for first-feeding Yellow Perch larvae has been identified as an influence on survival in Lake Michigan (Dettmers et al. 2003; Redman et al. 2011; Weber et al. 2011) and Canadian boreal lakes (Leclerc et al. 2011), and survival of European perch *Perca fluviatis* in the Baltic Sea (Ljunggren et al. 2003). The importance of adequate zooplankton supply and factors influencing zooplankton dynamics have been established for survival of Chesapeake Bay Striped Bass, White Perch, and American Shad larvae (North and Houde 2001; 2003; Hoffman et al. 2007; Martino and Houde 2010). Yellow Perch larvae share habitat in Chesapeake Bay subestuaries with these species, but little has been published on larval Yellow Perch dynamics and feeding ecology in Chesapeake Bay (Uphoff 1991).

Urbanization reduces quantity and quality of OM in streams (Paul and Meyer 2001; Gücker et al. 2011; Stanley et al. 2012). Riparian zones and floodplains that are sources of OM become disconnected from stream channels by stormwater management in suburban and urban watersheds (Craig et al. 2008; Kaushal et al. 2008; Brush 2009; NRC 2009). Small headwater streams in the Gunpowder and Patapsco rivers watersheds (tributaries of Chesapeake Bay) were sometimes buried in culverts and pipes, or were paved over (Elmore and Kaushal 2008). Decay of leaves occurred much faster in urban streams, apparently due to greater fragmentation from higher stormflow rather than biological activity (Paul and Meyer 2001). Altered flowpaths associated with urbanization affect the timing and delivery of OM to streams (McClain et al. 2003). Organic matter was transported further and retained less in urban streams (Paul and Meyer 2001). Uphoff et al. (2011a) found that the percentage of Maryland's Chesapeake Bay subestuary watersheds in wetlands declined hyperbolically as IS increased, so this source of OM diminished with development.

Management for OM (organic carbon) is nearly non-existent despite its role as a great modifier of the influence and consequence of other chemicals and processes in aquatic systems (Stanley et al. 2012). It is unmentioned in the Chesapeake Bay region as reductions in nutrients (N and P) and sediment are pursued for ecological restoration (http://www.epa.gov/reg3wapd/pdf/pdf_chesbay/BayTMDLFactSheet8_6.pdf). However, most watershed management and restoration practices have the potential to increase OM delivery and processing, although it is unclear how ecologically meaningful these changes may be. Stanley et al. (2012) recommended beginning with riparian protection or re-establishment and expand outward as opportunities permit. Wetland management represents an expansion of effort beyond the riparian zone (Stanley et al. 2012).

Agriculture also has the potential to alter OM dynamics within a watershed (Stanley et al. 2012) and the effect of this major land use on fish habitat warrants further study. Agriculture has been associated with increased, decreased, and undetectable changes in OM that may reflect diversity of farming practices (Stanley et al. 2012). As indicated earlier, extensive forest cover in a watershed may be linked to higher L_p than agriculture. However, Uphoff et al (2011a) noted that agricultural watersheds had more area in wetlands than urban watersheds and this could buffer loss of OM from decreased forest cover. Streams in agricultural watersheds were unlikely to become disconnected since urban stormwater controls would not be in use (Uphoff et al. 2011a).

In addition to feeding success, Yellow Perch egg viability declined greatly in highly developed suburban watersheds of Chesapeake Bay (Blazer et al. 2013). Abnormalities in ovaries and testes of adult Yellow Perch during spawning season were found most frequently in subestuaries with suburban watersheds and these abnormalities were consistent with contaminant effects (Blazer et al. 2013). Blazer et al. (2013) explained the biology behind low egg viability observed by Uphoff et al. (2005) in Severn River during 2001-2003 and persistently low L_p detected in three western shore subestuaries with highly developed suburban watersheds (C/ha > 1.59; Severn, South, and Magothy rivers). Endocrine disrupting chemicals were more likely to cause observed egg hatching failure in well developed tributaries than hypoxia and increased salinity (Blazer et al. 2013), factors identified as potential contributors to poor egg hatching success in Severn River (Uphoff et al. 2005). Low L_p occurs sporadically in subestuaries with rural watersheds and appears linked to high temperatures (Uphoff et al. 2013).

It is unlikely that low L_p has always existed in these suburban subestuaries since all supported productive and lightly regulated recreational fisheries into the 1970s (the C/ha threshold was met in Severn River during 1972). Severn River supported a state hatchery through the first half of the twentieth century and hatching rates of eggs in the hatchery were high into 1955, when records ended (Muncy 1962). Egg hatching success of Severn River Yellow Perch had declined drastically by the early 2000s when estimates of L_p were persistently low (Uphoff et al. 2005).

We used a general indicator of development (C/ha) in our analyses because negative effects of development involved multiple stressors difficult to isolate. Effects of multiple stressors are usually worse than the worst single stressor alone (Breitburg et al. 1998; Folt et al. 1999). Our results suggest a general sequence of stressors impacted yellow perch larvae as development increased. Feeding success declined as development proceeded past the target level of development and was followed by reduced egg hatching in highly developed subestuaries, implying initial stress related to disruption of OM dynamics followed by endocrine disrupting contaminants.

We did not interpret RNA/DNA ratios as rejecting or supporting the OM hypothesis since there was little indication of a match of zooplankton and Yellow Perch larvae in 2012 (primarily upper Bay subestuaries) or 2013 (primarily Potomac River subestuaries). Feeding success was much lower in these two years than during 2011. A contrasting year of high overall feeding success would greatly aid interpretation of RNA/DNA ratios. In a two-year study in Lake Saint

Pierre, Canada, Tardif et al. (2005) attributed larval Yellow Perch RNA/DNA response to wetland types, cumulative degree days, and feeding conditions.

Our RNA/DNA sampling indicated that most Yellow Perch larvae collected were in the starved category in both years (55 of 91 larvae in 2012 and 2013 (137 of 170). Larvae with an RNA/DNA ratio over 3 were detected more frequently in 2012 (14 of 91 larvae) than in 2013 (1 of 170 larvae). The response time of RNA/DNA ratios of larval fishes characterizes the feeding environment within a week of sampling (Tardif et al. 2005).

Ratios of RNA/DNA of fed larvae were expected to increase with body size (Clemmensen 1994). We observed an increase over the three sample periods during 2013 in Nanjemoy Creek, but not Mattawoman Creek during 2013 or in samples from 2012. Surveys of larval Striped Bass RNA/DNA in 1981 in the Potomac River estuary exhibited a similar declining pattern that we detected for Yellow Perch larvae, but Striped Bass ratios stabilized above starvation values (Martin et al. 1985). Blom et al. (1997) detected a decline in RNA/DNA ratios of Atlantic herring *Clupea harengus*; but few herring larvae were observed with ratios indicating starvation. Laboratory studies of RNA/DNA ratios of fed and starved larval Yellow Perch have not been conducted and we have relied on general guidelines from other species (Blom et al. 1997). Tardif et al. (2005) determined that RNA/DNA ratios of Yellow Perch in Lake Saint Pierre, Canada, averaged below 2, but did not provide indication of nutritional state of these larvae.

Low RNA/DNA ratios exhibited by some Yellow Perch at 7-9 mm may have reflected problems as they changed to external nutrition. RNA/DNA ratios of Atlantic Herring larvae fed shortly after hatching were in the same range as those found for starved larvae and were thought to result from the problems in changing from internal to external nutrition (Clemmenson 1994). There was no difference in RNA/DNA ratios for starved and fed Atlantic Herring larvae up to an age of 10 days. After 10 days, deprivation of food lead to a significant decrease in RNA/DNA ratios in comparison to fed Atlantic Herring larvae (Clemmensen 1994). Low RNA/DNA ratios of larger and presumably older Yellow Perch larvae sampled from our subestuaries may have been more indicative of poor feeding conditions, although it was possible that bias may have resulted from starving, weaker, poorly growing larvae being more vulnerable to our plankton nets than fed larvae.

In our analyses, we assumed that mainstem Potomac or Susquehanna River water was not a major influence on subestuary water quantity, water quality, and zooplankton supply. Sampling for Yellow Perch larvae occurred in the upper portions of subestuaries and this should have minimized the influence of mainstem waters, although some intrusion would have been possible at the most downstream sites in the smallest systems closest to the major rivers (i.e., Piscataway Creek for the Potomac). Strong correlations of C/ha, L_p, and OM0 indicated that local conditions prevailed.

Estimates of mean conductivity in subestuaries sampled during 2010-2011 (Uphoff et al. 2012) offered further evidence that local conditions were captured. Increases in stream conductivity have been strongly associated with urbanization (Wang and Yin 1997; Paul and Meyer 2001; Wenner et al. 2003; Morgan et al. 2007; Carlisle et al. 2010; Morgan et al. 2012) and have been noted in anadromous fish spawning streams in Maryland's portion of Chesapeake Bay (see Section 2-1). During 2010-2012, mean daily conductivities (219-249 μS/cm) in tidal-fresh Piscataway Creek's subestuary were elevated over those of tidal-fresh Mattawoman Creek's subestuary (range = 139-188 μS / cm) in spite of Piscataway Creek's upstream location on the Potomac River. In 2010, mean conductivities at two Chesapeake Bay Program monitoring stations corresponding to the mouths of Piscataway and Mattawoman Creeks averaged 211-212 μS / cm (once-monthly measurements at six depths during March and April; W. Romano, MD DNR, personal communication). Elevated conductivity in Piscataway Creek indicated that urbanization impacted estuarine water quality as well as stream water quality.

High estimates of L_p that were equal to or approaching 1.0 have been routinely encountered in the past, and it is likely that counts would be needed to measure relative abundance if greater resolution was desired. Mangel and Smith (1990) indicated that presence-absence sampling of eggs would be more useful for indicating the status of depleted stocks and count-based indices would be more accurate for recovered stocks. Larval indices based on counts have been used as a measure of year-class strength generally (Sammons and Bettoli 1998) and specifically for Yellow Perch (Anderson et al. 1998). Tighter budgets necessitate development of less costly indicators of larval survival and relative abundance in order to pursue ecosystem-based fisheries management. Characterizations of larval survival and relative abundance normally are derived from counts requiring labor-intensive sorting and processing. Estimates of L_p were largely derived in the field and only gut contents and RNA/DNA required laboratory analysis. These latter two analyses represented separate studies rather than a requirement for estimating L_p.

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Job 1 Section 3 - Estuarine Fish Community Sampling

Introduction

Water quality and aquatic habitat within watersheds is altered by agricultural activity and urbanization; both include use of pesticides and fertilizers, while the latter may have additional industrial wastes, contaminants, stormwater runoff and road salt (Brown 2000; NRC 2009; Benejam et al. 2010) that act as ecological stressors. Extended exposure to biological and environmental stressors affect fish condition and survival (Rice 2002; Barton et al. 2002; Benejam et al. 2008; Benejam et al. 2010). Reviews by Wheeler et al. (2005), the National Research Council (NRC 2009) and Hughes et al. (2014a; 2014b) documented deterioration of non-tidal stream habitat with urbanization.

Uphoff et al. (2011a) estimated target and limit impervious surface reference points (ISRPs) for productive fish habitat in brackish (mesohaline) Chesapeake Bay subestuaries based on Chesapeake Bay dissolved oxygen (DO) criteria, and associations and relationships of watershed impervious surface (IS), summer DO, and presence-absence of recreationally important finfish in bottom waters. Watersheds at a target of 5.5% IS (based on Towson University estimates for 1999-2000) or less (rural watershed) maintained mean bottom DO above 3.0 mg/L (threshold DO), but mean bottom DO was only occasionally at or above 5 mg/L (target DO). Mean bottom DO seldom exceeded 3.0 mg/L above 10% IS (suburban threshold; Uphoff et al. 2011a). Although bottom DO concentrations were influenced by development (indicated by IS) in brackish subestuaries, Uphoff et al. (2011b; 2012) have found adequate concentrations of DO in bottom channel habitat of tidal-fresh and oligohaline subestuaries with watersheds at suburban and urban levels of development. They suggested bottom channel waters were not succumbing to low oxygen because stratification due to salinity was weak, allowing them to become well mixed. However, the summer fish community of tidal-fresh Mattawoman Creek underwent drastic changes in abundance and species richness as development threshold was approached that were unrelated to adequacy of DO in channel waters, indicating other stressors were important (Uphoff et al. 2009; 2012).

In 2013, we continued to evaluate summer nursery and adult habitat for recreationally important finfish in tidal-fresh, oligohaline, and mesohaline subestuaries of Chesapeake Bay. In this report, we evaluated the influence of watershed development on target species presence-absence and abundance, total abundance of finfish, finfish species richness, and the probability that harvestable sized White Perch would be encountered by recreational fishermen. White perch are a popular estuarine panfish and, unlike many of the finfish we sample, sizes of interest to anglers are encountered regularly in our surveys. We continued to emphasize Mattawoman Creek in this report as part of Maryland DNRs' efforts to influence Charles County into modifying its comprehensive growth plan to conserve natural resources of its watershed (MDDNR 2013).

Figure 3-1. Subestuaries sampled in 2013, estuarine fish summer sampling.

Methods

We sampled nine subestaries in Chesapeake Bay during 2013: Broad Creek, Harris Creek, and Tred Avon River, tributaries of the Choptank River; Mattawoman Creek, Piscataway Creek, and Nanjemoy Creek, tributaries of the Potomac River; Northeast River, Middle River, and Gunpowder River located in the upper Chesapeake Bay (Table 3-1; Figure 3-1). This is the second year of sampling of Broad Creek and Harris Creek. These watersheds, downstream of Tred Avon River (sampled since 2006), represented a gradient of development from 0.29 C/ha (Broad Creek) to 0.75 C/ha (Tred Avon) within a single watershed (Table 3-1); Harris Creek is undergoing an

Region	Subestuary	IS (%)	C/ha	Salinity class	Watershed (ha)	Water (ha)
Mid-Bay	Broad Creek	5.1	0.29	Mesohaline	4,730	3,148
Mid-Bay	Harris Creek	6	0.39	Mesohaline	3,696	2,919
Mid-Bay	Middle River	23.4	3.33	Oligohaline	2,753	982
Mid-Bay	Tred Avon River	9.2	0.75	Mesohaline	9,563	2,429
Potomac	Mattawoman	10.4	0.91	Tidal-fresh	24,441	729
Potomac	Nanjemoy Creek	2.4	0.09	Oligohaline	18,893	1,131
Potomac	Piscataway Creek	13.9	1.46	Tidal-fresh	17,642	361
Upper	Bush River	14.1	1.48	Oligohaline	36,038	2,962
Upper Bay	Gunpowder River	9	0.72	Oligohaline	113,760	4,108
Upper	Northeast River	6.8	0.46	Tidal-fresh	16,342	1,579

Table 3-1. Percent impervious cover (IS), structures per hectare (C/ha), salinity class, watershed area, and area of tidal water for the subestuaries sampled.

extensive Oyster restoration effort (MD DNR 2014). Three Potomac River tributaries were sampled in 2013; Mattawoman Creek has been sampled since 1989, Piscataway Creek since 2006 (except in 2008), and Nanjemoy Creek since 2008 (NOAA's Integrated Assessment Project staff sampled with compatible methods in 2011-2012). Three subestuaries were sampled in upper Chesapeake Bay I 2013: Northeast River (sampled since 2007), Middle River (since 2009), and Gunpowder River (since 2009; Table 3-1).

We obtained compatible data from Bush River monitoring by citizen volunteers and staff from the Anita C. Leight Estuary Center (Bush River; Table 3-1; Figure 3-1). The Bush River has been sampled since 2006; the Estuary Center and its citizen volunteers, trained in 2011 by the Fisheries Service staff, have taken over sampling.

Housing density (C/ha) and impervious surface (IS) were estimated for each watershed (Table 3-1). We used property tax map based counts of structures in a watershed, standardized to hectares (C/ha), as our indicator of development (Uphoff et al. 2012). This indicator has been provided to us by Marek Topolski of the Fishery Management Planning and Fish Passage Program.

Tax maps are graphic representations of individual property boundaries and existing structures that help State tax assessors locate properties (Maryland Department of Planning or MDP 2013). All tax data were organized by county. Since watersheds straddle political boundaries, one statewide tax map was created for each year of available tax data, and then subdivided into watersheds. Maryland's tax maps are updated and maintained electronically as part of MDP's Geographic Information System's (GIS) database. Files were managed and geoprocessed in ArcGIS from Environmental Systems Research Institute (ESRI 2009). All feature datasets, feature classes, and shapefiles were spatially referenced using the NAD_1983_StatePlane_Maryland_FIPS_1900 projection to ensure accurate feature overlays and data extraction. ArcGIS geoprocessing models were developed using ArcGIS Model Builder to automate assembly of statewide tax maps, query tax map data, and assemble summary data. Each year's statewide tax map was clipped using the Maryland 8-digit watershed boundary file to create watershed land tax maps. Watershed area estimates excluded estuarine waters. These watershed tax maps were queried for all parcels having a structure built from 1700 to the tax data year. A large portion of parcels did not have any record of year built for structures but consistent undercounts should not have presented a problem since we were interested in the trend and not absolute magnitude (Uphoff et al. 2012).

Uphoff et al. (2012) developed a nonlinear regression equation to convert annual estimates of C/ha to estimates of percent impervious surface (IS) calculated by Towson University based on 1999-2000 (years in common) satellite imagery. The relationship of C/ha and IS was well described by the equation

$$IS = 10.98\ (C/ha)^{0.63}, (r^2 = 0.96; P < 0.0001).$$

Estimates of C/ha that were equivalent to 5% IS (target level of development for fisheries; a rural watershed), 10% IS (development threshold for a suburban watershed), and 15% IS (highly developed suburban watershed) were estimated as 0.27, 0.83, and 1.59 C/ha, respectively (Uphoff et al. 2012).

Tidal water surface area of each subestuary was estimated using the planimeter function on MDMerlin satellite photographs and maps (www.mdmerlin.net; Table 3-1). Shorelines were traced five-times for each system, and an average area was calculated. The lower limit of each water body was arbitrarily determined by drawing a straight line between the lowest downriver points on opposite shores (the mouth of each system) and the upper limits were to include all waters influenced by tides.

Surveys focused on eleven target species of finfish that fell within four broad life history groups: anadromous (American Shad, Alewife, Blueback Herring, Striped Bass), estuarine residents (White Perch, Yellow Perch), marine migrants (Atlantic Menhaden and Spot), and tidal-fresh forage (Spottail Shiner, Silvery Minnow, Gizzard Shad). With the exception of White

Perch, adults of the target species were rare and juveniles were common. Use of target species is widespread in studies of pollution and environmental conditions (Rice 2003). These species are widespread and support important recreational fisheries in Chesapeake Bay (directly or as forage); they are sampled well by commonly applied seine and-or trawl techniques (Bonzek et al. 2007); and the Bay serves as an important nursery for them (Lippson 1973; Funderburk et al. 1991). Gear specifications and techniques were selected to be compatible with other Fisheries Service surveys.

Ideally, four evenly spaced haul seine and bottom trawl sample sites were located in the upper two-thirds of each subestuary. Nanjemoy and Piscataway were covered sufficiently by three sites; however, in 2011 and 2012, NOAA added an additional site upstream of our three sites. Sites were not located near a subestuary's mouth to reduce influence of mainstem waters on fish habitat. We used GPS to record latitude and longitude at the middle of the trawl site, while latitude and longitude at seining sites were taken at the seine starting point on the beach.

Sites were sampled once every two weeks during July-September. All sites on one river were sampled on the same day, usually during morning through mid-afternoon. Sites were numbered from upstream (site 1) to downstream (site 4). The crew leader flipped a coin each day to determine whether to start upstream or downstream. This coin-flip somewhat randomized potential effects of location and time of day on catches and DO. However, sites located in the middle would not be as influenced by the random start location as much as sites on the extremes because of the bus-route nature of the sampling design. If certain sites needed to be sampled on a given tide then the crew leader deviated from the sample route to accommodate this need. Trawl sites were generally in the channel, adjacent to seine sites. At some sites, seine hauls could not be made because of permanent obstructions, SAV beds, or lack of beaches.

Water quality parameters were recorded at all sites. Temperature (ºC), DO (mg/L), conductivity (mS/cm), salinity (‰), and pH were recorded at the surface, middle, and bottom of the water column at the trawl sites and at the surface of the seine site. Mid-depth measurements were omitted at sites with less than 1.0 m difference between surface and bottom. Secchi depth was measured to the nearest 0.1 m at each trawl site. Weather, tide state (flood, ebb, high or low slack), date, and start time were recorded for all sites.

Dissolved oxygen concentrations were evaluated by watershed against a target of 5.0 mg/L and a threshold of 3.0 mg/L (Batiuk et al. 2009; Uphoff et al. 2011a). This target DO is considered sufficient to support aquatic life needs in Chesapeake Bay (Batiuk et al. 2009) and has been used in a regulatory framework to determine if a water body is meeting its designated aquatic life uses. The target criterion was associated with asymptotically high presence of target species in bottom channel habitat in brackish subestuaries (Uphoff et al. 2011a). Presence of target species declined sharply when bottom DO fell below the 3.0 mg/L threshold (Uphoff et al. 2011a). In each subestuary, we estimated the percentages of DO samples that did not meet the target or threshold for all samples (surface to bottom) and for bottom waters alone. The percentages of DO measurements that met or fell below the 5 mg/L target (V_{target}) or fell at or below the 3 mg/L threshold ($V_{threshold}$) were estimated as

$$V_{target} = (N_{target} / N_{total})*100;$$

and

$$V_{threshold} = (N_{threshold} / N_{total})*100;$$

where N_{target} was the number of measurements meeting or falling below 5 mg/L, $N_{threshold}$ was the number of measurements falling at or below 3 mg/L, and N_{total} was total sample size.

Conductivity measurements were collected at each site in every system from July to September. Conductivity measurements recorded in 2012-2013 were recorded incorrectly. The raw conductivity was recorded instead of the specific conductivity, which compensates for temperature. An equation was used to correct the error and convert the raw conductivity measurements that were recorded to specific conductivity (Fofonoff and Millard 1983):

$$\text{Specific Conductivity} = \text{Conductivity} / (1 + ((0.02 \cdot T) - 25));$$

for each ºC change in water temperature (T) there was a 2% change in conductivity.

Each subestuary was classified into a salinity category based on the Venice System for Classification of Marine Waters (Oertli, 1964). Salinity influences distribution and abundance of fish (Hopkins and Cech, 2003; Cyrus and Blaber, 1992; Allen, 1982) and DO (Kemp et al. 2005). We calculated an arithmetic mean of all bottom salinity measurements over all years available to determine salinity class for each subestuary (Uphoff et al. 2012). Tidal-fresh ranged from 0-0.5 ‰; oligohaline, 0.5-5.0 ‰; and meshohaline, 5.0-18.0 ‰ (Oertli, 1964). Mattawoman Creek, Piscataway Creek, and Northeast River were classified as tidal-fresh subestuaries (Table 3-1). Gunpowder River, Bush River, Middle River, and Nanjemoy Creek were considered oligohaline. Broad Creek, Harris Creek, and Tred Avon River, were mesohaline subestuaries (Table 3-1). We grouped data by these classifications when examining effects of development.

A 4.9 m headrope semi-balloon otter trawl was used to sample fish in mid-channel bottom habitat. The trawl was constructed of treated nylon mesh netting measuring 38 mm stretch-mesh in the body and 33 mm stretch-mesh in the codend, with an untreated 12 mm stretch-mesh knotless mesh liner. The headrope was equipped with floats and the footrope was equipped with a 3.2 mm chain. The net used 0.61 m long by 0.30 m high trawl doors attached to a 6.1 m bridle leading to a 24.4 m towrope. Trawls were towed in the same direction as the tide. The trawl was set up tide to pass the site halfway through the tow, allowing the same general area to be sampled regardless of tide direction. A single tow was made for six minutes at 3.2 km / hr (2.0 miles / hr) per site on each visit. The contents of the trawl were emptied into a tub for processing.

During 2009-2013, a 3.1 m box trawl made of 12.7 mm stretch-mesh nylon towed for five minutes was used on the same day sampling was conducted with a 4.9 m trawl in Mattawoman Creek to create a catch-effort time-series directly comparable to monitoring conducted during 1989-2002 (Carmichael et al. 1992). The initial choice of net to start with on each day in Mattawoman Creek was decided by a coin flip.

An untreated 30.5 m 1.2 m bagless knotted 6.4 mm stretch mesh beach seine, the standard gear for Bay inshore fish surveys (Carmichael et al. 1992; Durell 2007), was used to sample inshore habitat. The float-line was rigged with 38.1 mm by 66 mm floats spaced at 0.61 m intervals and the lead-line rigged with 57 gm lead weights spaced evenly at 0.55 m intervals. One end of the seine was held on shore, while the other was stretched perpendicular to shore as far as depth permitted and then pulled with the tide in a quarter-arc. The open end of the net was moved towards shore once the net was stretched to its maximum. When both ends of the net were on shore, the net was retrieved by hand in a diminishing arc until the net was entirely pursed. The section of the net containing the fish was then placed in a washtub for processing. The distance the net was stretched from shore, maximum depth of the seine haul, primary and secondary bottom type, and percent of seine area containing aquatic vegetation were recorded.

All fish captured were identified to species and counted. Striped Bass and Yellow Perch were separated into juveniles and adults. White Perch were separated into three categories (juvenile, small adults and harvestable size) based on size and life stage. The small adult White Perch category consisted of ages-1+ White Perch smaller than 200 mm. White Perch greater than or equal to 200 mm were considered to be of harvestable size and all captured were measured to the nearest millimeter. White Perch of this size or larger corresponded to the quality length category minimum (36-41% of the world record TL) proposed by Anderson (1980) for proportional stock density (PSD) indices; 200 mm TL is used as the length cut-off for White Perch in Chesapeake Bay assessments of White Perch (Piavis and Webb 2013). Small adult and harvestable White Perch were combined when catches were summarized as adults. Catch data were summarized and catch statistics were reported for both gears combined and each gear separately.

Three basic metrics of community composition were estimated for subestuaries sampled: geometric mean catch of all species, total number of species (species richness), and species comprising 90% of the catch. The geometric mean (GM) was estimated as the back-transformed mean of $\log_e$-transformed catches (Ricker 1975; Hubert and Fabrizio 2007). Prior to this report, only the arithmetic mean of catches (AM) was reported. The GM is a more precise estimate of central tendency of fish catches than the AM, but is on a different scale than the AM (Ricker 1975; Hubert and Fabrizio 2007). Both values are given in this report for comparison to prior reports and for future reports. We noted which target species were within the group comprised 90% of fish collected. We summarized these metrics by salinity type since some important ecological attributes (DO and high or low SAV densities) appeared to reflect salinity class (Uphoff et al. 2012).

We plotted number of species collected by seine or 4.9 m trawl against C/ha and denoted salinity class on these two plots. A greater range of years (1989-2012) was available for seine samples than the 4.9 m trawl (2003-2012) due to a change from the 3.1 m trawl used during 1989-2002 (Carmichael et al. 1992). This was an exploratory analysis because not all subestuaries and years had C/ha estimates. The same plot was constructed for GMs of total catch. These plots would provide insight on how salinity class and C/ha influenced species richness and total abundance. We set a minimum number of samples (15) for a subestuary in a year to include estimates of species richness. This eliminated years where sampling in a subestuary had to be ended due to site losses typically from SAV growth that did not permit sampling throughout a season. We plotted the total number of species by their respective number of trawl or seine samples collected to see if we could detect an influence of sample size on accumulation of species (Kwak and Peterson 2007). If a linear or non-linear relationship of richness was suggested, a suitable regression was run. If significant at $\alpha \leq 0.05$, the residuals were used as an effort-corrected time-series of relative richness (above or below average, with the average indicated by 0) plotted against C/ha to examine whether a trend in species richness might be suggested.

We discovered an error in Nanjemoy Creek data during 2011-2012 due to station identifications being switched; stations were correctly identified using the latitudes-longitudes recorded at the time of sampling and matched to the correct sites. Errors were also discovered in Mattawoman 2009-2011 trawl data, the 3.1 m box trawls and 4.9 m headrope semi-balloon otter trawls were not correctly identified; trawls were correctly identified by going through field data sheets.

Individual total lengths (TL) of White Perch (>200 mm TL) that should be of interest to anglers have been collected during trawl and seine sampling since 2004. White Perch of this size or larger corresponded to the quality length category minimum (36-41% of the world record TL) proposed by Anderson (1980) for proportional stock density (PSD) indices; 200 mm TL is used as the length cut-off for White Perch in Chesapeake Bay stock density indices (Piavis and Webb 2013). These data provided an opportunity to evaluate the influence of development on the availability of fish for anglers to harvest.

Annual proportions of seine or trawl samples in a subestuary with quality length or greater (≥ 200mm) White Perch and their 95% CI were calculated. The proportion of samples with quality length or greater White Perch was estimated as

$$P_{Qwp} = N_{Quality} / N_{total};$$

where $N_{Quality}$ equaled the number of samples with quality length or greater White Perch present and N_{total} equaled the total number of samples taken. The SD of each P_{Qwp} was estimated as

$$SD = [(P_{Qwp}(1 - P_{Qwp})) / N_{total}]^{0.5} \text{ (Ott 1977).}$$

The 95% confidence intervals were constructed as

$$P_{Qwp} \pm (1.96\ SD).$$

Two approaches were considered for examining the effect of development (C/ha) on the availability of quality size White Perch (P_{Qwp}): linear regression and probability of encounter within development categories delineated by our C/ha target and threshold. We examined histograms of frequency of P_{Qwp} in seine and trawl samples and decided to pool P_{Qwp} estimates by gear into one analysis, i.e., seine- and trawl-based estimates of P_{Qwp} were plotted against C/ha on the same graph and were not treated separately. Salinity classes and gear were denoted by different symbols on the plot. Wide scatter of P_{Qwp} (0 - 0.79) for both gears at lower C/ha indicated that even a significant regression was unlikely to explain enough variation to be useful to management, but it would detect the trend. It should be noted that trawl-based estimates of P_{Qwp} described availability in bottom channel habitat, while seine-based estimates of P_{Qwp} described availability in shallow, shore zone habitat.

We used the upper and lower quartiles of all (seine and trawl samples together) P_{Qwp} to define "good" and "poor" opportunities, respectively, for fishermen to encounter harvestable White Perch. We then determined the proportions of estimates of P_{Qwp} that were in the upper (P_{75th}) or lower quartiles (P_{25th}) for C/ha at or less than 0.27 (rural watershed and target condition), greater than 0.83 C/ha (suburb to urban watershed and threshold or worse condition), and in between (transition watershed). Estimators of P_{75th} or P_{25th}, their SD's, and 95% CI's were constructed as described above for P_{Qwp}, with number of estimates within a development class that were in the upper or lower quartiles substituted for $N_{Quality}$ and total number of estimates within a development class substituted for N_{total} Overlap of 95% CI's was used to determine whether estimates of P_{75th} or P_{25th} were different among the three levels of development.

Presence-absence was ecologically meaningful, minimized errors and biases in sampling, and reduced statistical concerns about lack of normality and high frequency of zero catches that were expected given the hypothesis that increased development leads to reduced habitat suitability (Green 1979; Bannerot and Austin 1983; Mangel and Smith 1990; Uphoff et al. 2011a).

We continued to track bottom DO, SAV area, finfish abundance and finfish species richness in 3.1 m and 4.9 m trawl samples from Mattawoman Creek and compared them to changes in C/ha.

We obtained measurements of total ammonia nitrogen (TAN; NH_3 plus NH_4; US EPA 2013) in Mattawoman Creek during the SAV growing season (April-October) from a Chesapeake Bay Program (CBP) monitoring site MAT0016 located in the channel between our stations 3 and 4 (W. Romano, MD DNR, personal communication). Estimates were available for 1986-2012, but we eliminated 1986-1990 from analysis because of methodology differences. Measurements of growing season TAN were annually summarized as minimum, median, and maximum and compared to US EPA ambient water quality criteria for TAN (US EPA 2013) to capture the potential for acute and chronic toxicity.

Sampling with 3.1 m trawls was conducted during 1989-2002 and 2009-2013 and 4.9 m trawls have been used since 2003. Geometric means and their 95% CI's of total fish abundance were estimated for the 3.1 m trawl for samples from Mattawoman Creek. When we compared trends of GMs of total fish abundance in the years in common for the 3.1 m and 4.9 m trawls in Mattawoman Creek we noted a close correspondence. We decided to develop a linear regression of 4.9 m and 3.1 m GMs to predict the missing portion (2003-2008) of the 3.1 m GM time-series.

Estimates of species richness in Mattawoman Creek (number of species encountered) were made for 3.1 m trawl samples during 1989-2002 and 2009-2013. Sampling during 1989-2002 was based on monthly sam-

Subestuary	Salinity Classification	C/ha	All DO % < 5.0 mg/L	Bottom DO % < 5.0 mg/L	Bottom DO % < 3.0 mg/L
Broad Creek	Mesohaline	0.29	0.1	0.3	0
Harris Creek	Mesohaline	0.39	0	0	0
Tred Avon River	Mesohaline	0.75	0.15	0.35	0.15
Middle River	Oligohaline	3.33	0.06	0.18	0
Gunpowder River	Oligohaline	0.72	0.04	0.17	0
Nanjemoy Creek	Oligohaline	0.09	0.09	0.25	0
Bush River	Oligohaline	1.48	0.09	0.17	0
Mattawoman	Tidal Fresh	0.91	0	0	0
Northeast River	Tidal Fresh	0.46	0	0	0
Piscataway Creek	Tidal Fresh	1.46	0	0	0

Table 3-2. Percentages of all DO measurements and bottom DO measurements that did not meet target (≤ 5.0 mg/L) and threshold (≤ 3.0 mg/L) conditions during July-September, 2013, for each subeastuary. C/ha = structures per hectare.

pling of 5 stations (Carmichael et al. 1992). The outermost station sampled during 1989-2002 was outside the range of those sampled during 2009-2013 and this station was dropped. Remaining stations were the same throughout the time-series, but were sampled monthly during 1989-2002 (annual sample size = 12) and bi-monthly during 2009-2013 (annual sample size = 24). In order to match the annual sample sizes of 1989-2002, we made two sets of estimates for each sample year during 2009-2012: one for the first round of the month and one for the second. As a result, all comparisons of species richness in Mattawoman Creek were based on the same annual sample size.

Results and Discussion

Harris Creek, and the three tidal-fresh subestuaries did not have DO readings less than the target level (5.0 mg/L) during 2013 (Table 3-2). Remaining subestuaries had non-zero estimates of V_{target} in surface and bottom waters. Fifteen percent of DO measurements from Tred Avon River were below the target (V_{target} = 15%); Broad Creek, 10%; Nanjemoy Creek, 9%; Bush River, 9%; Middle River, 6%; and Gunpowder River, 4%. When we evaluated V_{target} in bottom channel waters, Tred Avon River had the highest estimate at 35%; followed by Broad Creek, 30%; Nanjemoy Creek, 25%; Middle River, 18%; and Gunpowder and Bush rivers, 17%; all other subestuaries had V_{target} estimates of zero. Only Tred Avon River had measurements of bottom DO below the 3 mg/L threshold during 2013 (Table 3-3); these occurred at the uppermost site closest to Easton.

DO Depth	Statistics	Temperature Depth	C/ha
Mesohaline			
Surface	r	-0.144	0.016
	α	0.31	0.913
	N	52	51
Bottom	r	-0.082	-0.594
	α	0.563	<0.0001
	N	52	51
Oligohaline			
Surface	r	-0.388	0.338
	α	0.145	0.085
	N	27	27
Bottom	r	-0.396	-0.16
	α	0.41	0.413
	N	27	27
Tidal-fresh			
Surface	r	0.013	0.32
	α	0.95	0.11
	N	26	26
Bottom	r	0.101	0.4
	α	0.623	0.043
	N	26	26

Table 3-3. Correlations of 2003-2013 arithmetic mean annual surface and bottom DO (mg/L) with like water temperatures at depth (surface and bottom) or watershed development (C/ha = structures per hectare), by salinity class.

Subestuary	Stations Sampled	Number of Samples	Species	Species Comprising 90% of Catch	C/ha	Total Catch	AM	GM
				Atlantic Silveside				
Broad Creek	3	18	20	Striped Killifish	0.29	3889	216.1	176.8
				Gizzard Shad				
				White Perch				
				Spottail Shiner				
				YOY White Perch				
Bush River	4	16	29	Atlantic Menhaden	1.48	4447	277.9	226.2
				Gizzard Shad				
				White Perch				
				Spottail Shiner				
				Atlantic Menhaden				
				Bay Anchovy				
				Pumpkinseed				
				Atlantic Silverside				
				YOY White Perch				
Gunpowder River	3	17	26	Inland Silverside	0.72	1863	109.6	59.8
				Atlantic Silverside				
Harris Creek	3	17	20	Atlantic Menhaden	0.39	3526	207.4	198.1
				Pumpkinseed				
				Gizzard Shad				
				Atlantic Silverside				
				White Perch				
				Banded Killifish				
				YOY White Perch				
				Inland Silverside				
Middle River	2	8	23	Bluegill	3.33	953	119.1	96.8
				Atlantic Silverside				
				Bay Anchovy				
				YOY White Perch				
				White Perch				
				Mummichog				
Nanjemoy Creek	3	18	27	Inland Silverside	0.09	2030	112.8	96.3
				Gizzard Shad				
				Threadfin Shad				
Northeast River	4	24	23	Bay Anchovy	0.46	5755	239.8	148.5
				Atlantic Silverside				
				Bay Anchovy				
				White Perch				
				Atlantic Menhaden				
Tred Avon River	4	26	20	Striped Killifish	0.75	2934	112.8	94.8

Table 3-4. Beach seine catch summary, 2013. C/ha = structures per hectare. AM is the arithmetic mean catch of all fish per seine. GM is the geometric mean catch of all fish per seine.

Correlation analyses of 2003-2013 data suggested that the sign and significance ($\alpha \leq 0.05$) of associations of mean surface or bottom DO with C/ha were influenced by salinity classification in a manner consistent with potential for stratification (Table 3-3). In mesohaline subestuaries, where strongest stratification was expected, the association between bottom DO with C/ha was negative and significant ($r = -0.59$, $\alpha < 0.0001$), while remaining comparisons at the mesohaline and other salinity classifications were not. Given that multiple comparisons were made, the positive correlation of bottom DO with C/ha for tidal-fresh subestuaries ($\alpha = 0.04$) was considered spurious (Nakagawa 2004). Sample sizes of mesohaline subestuaries (N = 52) were over twice as high as oligohaline (N = 27) or tidal-fresh subestuaries (N = 26), so ability to detect significant associations in mesohaline subestuaries was greater (Table 3-3).

A total of 25,397 fish representing 48 species were captured by beach seine in 2013 (Table 3-4). Seven species comprised 90% of the total fish caught in 2013, including Atlantic Silverside (32%), Gizzard Shad (26%), Bay Anchovy (9%), White Perch (adult) (7%), Threadfin Shad (5%), White Perch (juvenile) (4%), Atlantic Menhaden (4%), and Striped Killifish (3%). White Perch (adults and juveniles), Gizzard Shad, and Atlantic Menhaden represented three target species among the species comprising 90% of the total catch (Table 3-4).

During 2013, dense submerged aquatic vegetation (SAV) prevented seining in Mattawoman and Piscataway Creeks. Seining in Middle River was sporadic because of high tides that limited beach availability and dense SAV in seine sites; only two seine sites were available when tide and SAV allowed. Seine sites sampled in Middle River and Nanjemoy Creek for NOAA's Integrated Assessment were dropped since NOAA terminated field collections. In Gunpowder River, one seine site (Site 2) was not sampled at all after it was roped off for swimming.

Geometric mean seine catches ranged from 59.8 to 226.2 during 2013, with little indication that salinity class or development level exerted an influence (Table 3-4). Interestingly, two adjacent oligohaline subestuaries had the highest and lowest GMs. Number of species estimated for Middle River (8 seine hauls) was excluded from analysis. Remaining subestuaries had 16-26 samples. Oligohaline subestuaries had the greatest number of species (26-29) during 2013. Twenty-three species were caught in the lone tidal-fresh subestuary that could be seined (Northeast River). Twenty species were collected from the three mesohaline subestuaries (each; Table 3-4).

Four target species were present among species comprising 90% of the seine catch (dominant species), White Perch (adults and Juvenile) in five of the eight subestuaries seined, Atlantic Menhaden in four, Gizzard Shad in three, and Spottail Shiner in two (Table 3-4). These species were frequently encountered in oligohaline subestuaries. Four of five subestuaries with White Perch (adults and Juvenile) comprising 90% of the catch were oligohaline. All three subestuaries where Gizzard Shad were observed as dominant species were oligohaline, as were two subestuaries with Atlantic Menhaden or Spottail Shiner comprising 90% of the catch. Remaining target species were not estimated among dominant species collected by seine (Table 3-4).

Bottom trawling with a 4.9 m headrope trawl was conducted in all ten subestuaries in 2013. A total of 65,626 fish and 37 fish species were captured (Table 3-5). Three species comprised 90% of the total catch for 2013, White Perch juvenile (35%), Bay Anchovy (32%), White Perch adults (12%), and Spottail Shiner (11%). White Perch (juveniles and adults) and Spottail Shiner were target species (Table 3-5).

Geometric mean trawl catches during 2013 were between 53.8 and 576.1 (Table 3-5). Mesohaline subestuaries had the lowest trawl GMs (53.8-137.1), while tidal-fresh subestuary GMs (184.9-286.8) were overlapped by oligohaline subestuary GMs (147.5-576.1). Number of species estimated for Bush River (based on 12 trawls) was excluded from analysis. Remaining subestuaries had 17-26 samples. Number of species captured by trawl in subestuaries sampled during 2013 (16-27) overlapped for all three salinity classifications (Table 3-5).

White Perch (juveniles and/or adults) were among species comprising 90% of 4.9 m trawl catches in every subestuary (Table 3-5). Bay Anchovy were the most frequently collected species in mesohaline subestuaries. Nanjemoy Creek had the highest total catch at 16,807 (933.7 fish per trawl, respectively) and Bush had the lowest total catch at 2,904 (242.0 fish per trawl, respectively). Nanjemoy Creek had the highest GM (576.1) and the Tred Avon River had the lowest GM (53.8; Table 3-5).

Species richness in seine collections made from mesohaline subestuaries (that met the effort criterion) appeared to be influenced by effort, while bivariate plots did not suggest a relationship for tidal-fresh or oligohaline subestuaries (Figure 3-2). Plots of species richness and C/ha did not suggest a relationship in tidal-fresh or oligohaline subestuaries (Figure 3-3). Tidal-fresh subestuary watersheds were represented by a limited range of C/ha (0.43-0.72) that fell between the rural watershed target and suburban threshold. Oligohaline subestuary watersheds were represented by the widest range of C /ha (0.09-3.33, rural to urban) of the three salinity classes (Figure 3-3).

Subestuary	Stations Sampled	Number of Samples	Species	Species Comprising 90% of Catch	C/ha	Total Catch	AM	GM
Broad Creek	4	24	19	Bay Anchovy	0.29	4718	197	137.1
Bush River	3	12	18	White Perch	1.48	2904	242	216.1
				Gizzard Shad				
				YOY White Perch				
				Bay Anchovy				
Gunpowder River	4	24	27	Bay Anchovy	0.72	4110	171	147.5
				YOY White Perch				
				White Perch				
				Spottail Shiner				
				Channel Catfish				
				Pumpkinseed				
				Brown Bullhead				
Harris Creek	4	24	16	Bay Anchovy	0.39	3992	166	72.4
				White Perch				
Mattawoman Creek	4	24	22	Spottail Shiner	0.91	11832	493	286.8
				YOY White Perch				
Middle River	4	24	19	Bay Anchovy	3.33	7618	317	182.6
				White Perch				
				YOY White Perch				
Nanjemoy Creek	3	18	21	YOY White Perch	0.09	16807	934	576.1
				Bay Anchovy				
Northeast River	4	24	17	White Perch	0.46	5511	230	187.3
				YOY White Perch				
				Brown Bullhead				
Piscataway Creek	3	17	20	YOY White Perch	1.46	3881	228	184.9
				Spottail Shiner				
				Tessellated Darter				
Tred Avon River	4	26	16	Bay Anchovy	0.75	4253	164	53.8
				White Perch				
				Hogchoker				

Figure 3-5. Bottom trawl (4.9m) catch summary, 2013. C/ha = structures per hectare. AM is the arithmetic mean catch of all fish per seine. GM is the geometric mean catch of all fish per seine.

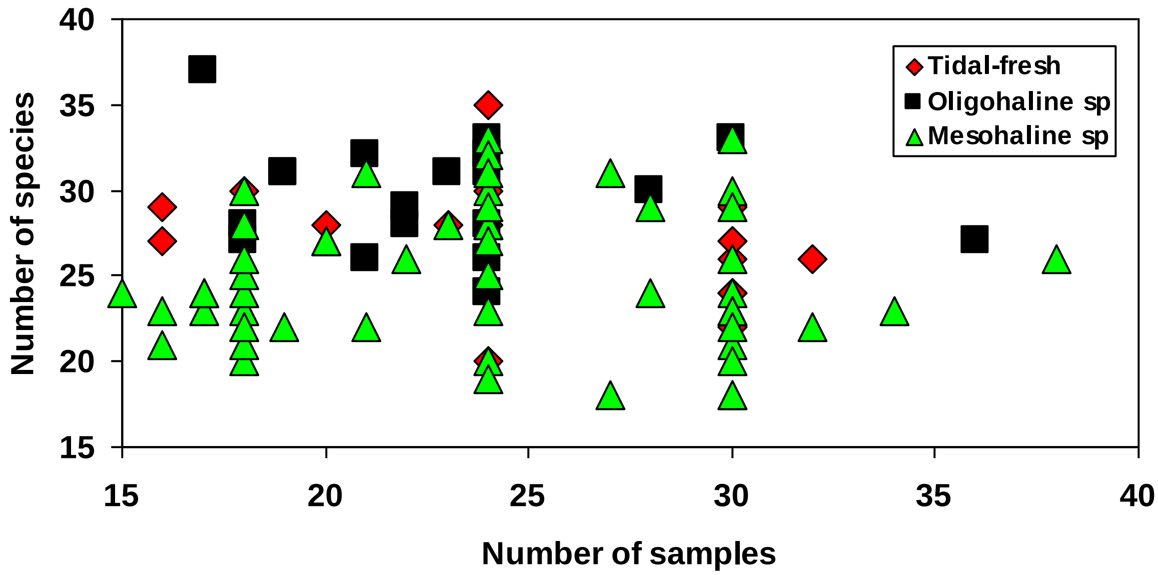

Figure 3-2. Number of species captured annually during 2003-2013 in subestuaries by seining plotted against number of seine samples taken, by salinity class.

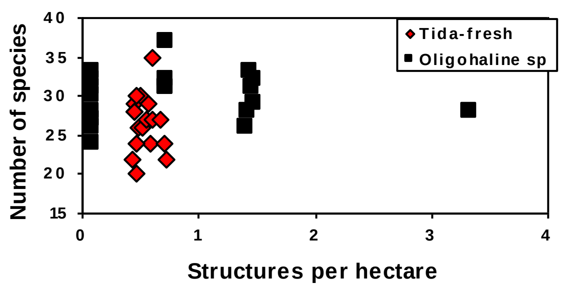

Figure 3-3. Number of finfish species collected by seining in fresh-tidal or oligohaline subestuaries versus intensity of watershed development (C/ha = structures per hectare).

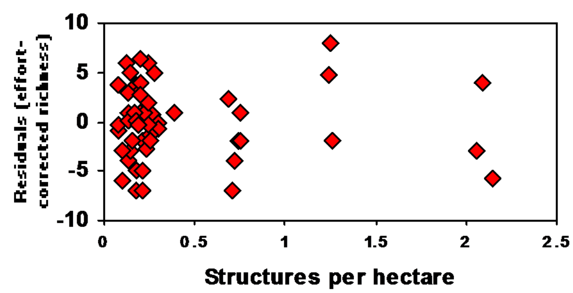

Figure 3-4. Residuals of regression of number of finfish species collected in mesohaline subestuaries and effort versus intensity of development (C/ha = structures per hectare). Residuals indicated trend for effort-corrected number of species (richness) was not related to development level.

High scatter of the relationship of species richness and seine effort in mesohaline subestuaries made selection of a suitable function (linear or nonlinear asymptotic) difficult. We selected a linear regression because of its minimal parameterization. The linear regression was significant (P = 0.00023, N = 54) but explained only 17% of variation. A plot of residuals against effort did not suggest bias from using a linear relationship; these residuals were used as an effort-corrected depiction of the trend in species richness with development (Figure 3-4). These residuals did not indicate an influence of C/ha on number of species collected from mesohaline subestuaries (Figure 3-4).

Similar to what was found with seine samples, species richness in 4.9 m trawl collections from mesohaline subestuaries that met the effort criterion appeared to be influenced by effort (Figure 3-5). Bivariate plots did not suggest a relationship for tidal-fresh or oligohaline subestuaries (Figure 3-6). Again, high scatter of the relationship of species richness and 4.9 m trawl effort in mesohaline subestuaries made selection of a suitable function difficult and we selected a linear regression. This linear regression was significant (P = 0.003, N = 37), but explained only 22% of variation.

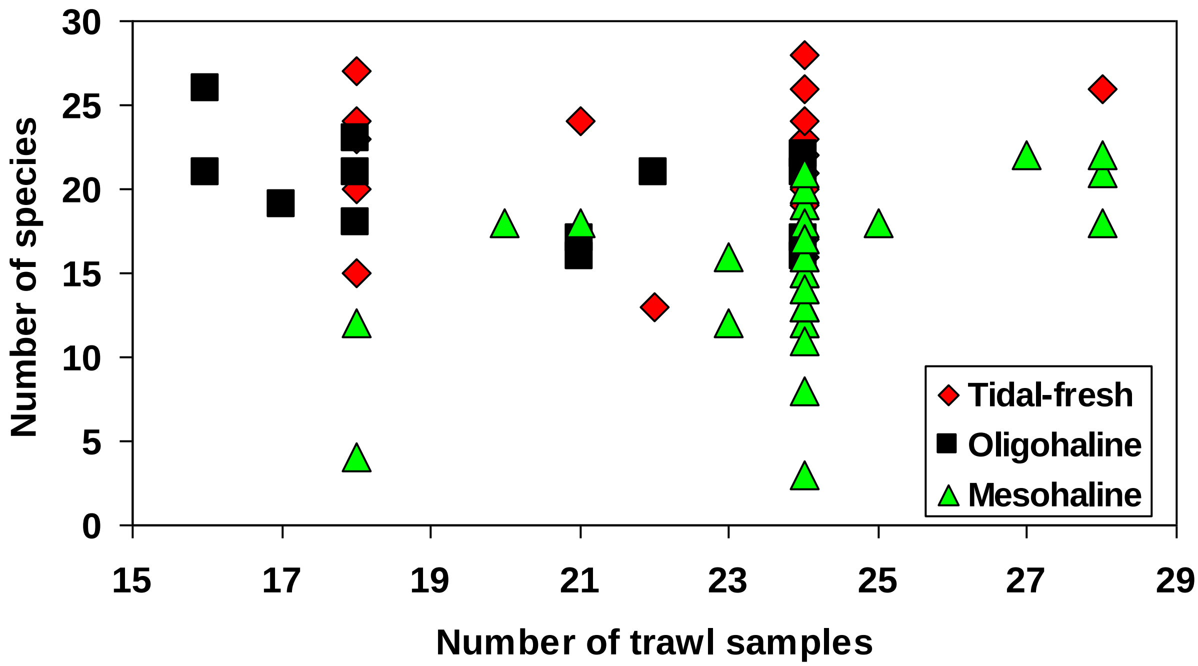

Figure 3-5. Number of species collected by 4.9 m trawl and sample size for tidal-fresh, oligohaline, and mesohaline subestuaries during 2003-2013.

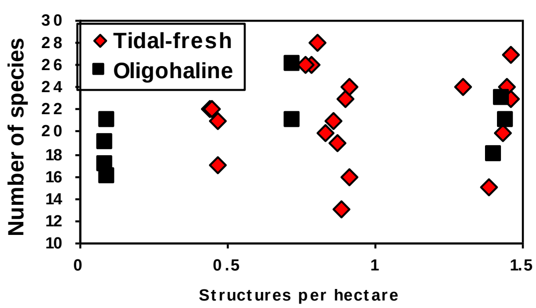

Figure 3-6. Number of finfish species collected by 4.9 m trawl in fresh-tidal or oligohaline subestuaries versus intensity of development (C/ ha = structures per hectare).

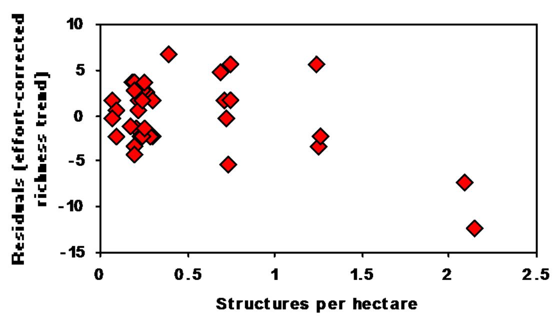

Figure 3-7. Residuals of regression of number of finfish species collected in mesohaline subestuaries and 4.9 m trawl effort versus intensity of development (C/ha = structures per hectare). Residuals indicated trend for effort-corrected number of species (richness) declined with beyond threshold development.

A plot of residuals against effort did not suggest bias from using a linear relationship; these residuals were used as an effort-corrected depiction of the trend in species richness with development.

Plots of species richness against C/ha in 4.9 m trawl collections did not suggest relationships for either tidal -fresh or oligohaline subestuaries (Figure 3-7). For mesohaline subestuaries, the plot of the residuals of the relationship of species richness with 4.9 m trawl effort versus C/ha suggested that the number of species declined when development went beyond the threshold (Figure 3-7).

In general these exploratory analyses of species richness and development supported trends found in analyses of development and DO. Levels of DO were not negatively influenced by development in tidal-fresh or oligohaline subestuaries, but were in mesohaline subestuaries. Depletion of DO in mesohaline subestuaries to hypoxic or anoxic levels represented a direct loss of habitat to be occupied. Uphoff et al. (2011) determined that the odds of target species (same as in this report, less the tidal-fresh forage component) being present in seine samples from mesohaline subestuaries were not influenced by development (indicated by percent impervious surface), but odds of target species being present in bottom channel trawl samples were negatively influenced by development. The extent of bottom channel habitat that can be occupied does not appear to diminish with development in tidal-fresh and oligohaline subestuaries. However, sampling of DO in dense SAV beds in tidal-fresh Mattawoman Creek in 2011 indicated that shallow water habitat could be negatively impacted by low DO within the beds (Uphoff et al. 2012); it was not feasible for us to sample fish within the beds so the impact on target finfish could not be estimated.

	Development		
Parameter	Rural	Between	Suburb
N	48	62	49
		25th Percentile	
N25th	7	10	26
P25	0.15	0.16	0.53
SD	0.05	0.05	0.07
Lower 95%	0.05	0.07	0.39
Upper 95%	0.25	0.26	0.63
		75th Percentile	
N75th	18	21	2
P75	0.38	0.34	0.04
SD	0.07	0.06	0.03
Lower 95%	0.24	0.22	-0.01
Upper 95%	0.51	0.46	0.1

Table 3-6. Summary of information used to determine the proportions of White Perch $\leq$ 200 mm, TL (quality sized) indicative of poor (25th percentile) and good (75th percentile) size availability to recreational fishermen. N = number of estimates of P_{Qwp} available, N25th = the number of samples within the 25th percentile; P25 is the proportion of samples in the 25th percentile; SD = the standard deviation; Lower 95% is the lower 95% confidence interval, and Upper 95% is the upper 95% confidence interval. Remaining abbreviations with 75 instead of 25 refer to

A total of 159 seine and trawl estimates of P_{Qwp} were available; 52% of estimates were from mesohaline subestuaries, 28% were from oligohaline subestuaries, and 19% were from tidal-fresh (Table 3-6; Figure 3-8). The upper quartile of all estimates of P_{Qwp} equaled 0.29 and the lower quartile contained 0 (0 accounted for 27% of the estimates of P_{Qwp}). Distribution of upper quartile estimates of P_{Qwp} among salinity types was dissimilar to the distribution of all seine and trawl estimates; 80% of 41 estimates within the upper quartile were from mesohaline subestuaries, and 10% each were from oligohaline and tidal-fresh subestuaries. Forty estimates of P_{Qwp} equaled 0 in all subestuaries; 16 were from mesohaline subestuaries, 10 equaled 0 in oligohaline subuestuaries, and 14 in tidal-fresh subestuaries. White Perch at a size of interest to anglers were more likely to be found in mesohaline subestuaries we have surveyed and least likely to be found in tidal-fresh subestuaries (Table 3-6; Figure 3-8).

The linear regression of P_{Qwp} against C/ha was significant and indicated a negative relationship overall, but explained little variation ($r^2 = 0.10$, $P < 0.0001$, N = 159). Thirty eight percent of estimates of P_{Qwp} in rural watersheds (C/ha $\leq$ 0.27) were within the upper quartile, while only 4% were within the upper quartile in suburban-urban watersheds (C/ha $\geq$ 0.83; Table 3-6; Figure 3-8). The percentage of P_{Qwp} estimates in the upper quartile that occurred as C/ha made a transition from rural to suburban, 34%, was similar to that of a rural watershed. Based on 95% CI overlap, the percentages of P_{Qwp} estimates in the upper quartile were not significantly different between rural and transition watersheds, but both were greater than that for suburban-urban watersheds. Fifteen percent of estimates of P_{Qwp} in rural watersheds (C/ha $\leq$ 0.27) equaled 0, while 53% equaled zero in suburban-urban watersheds (C/ha $\geq$ 0.83). The percentage of P_{Qwp} estimates equal to zero that occurred as C/ha made a transition from rural to suburban, 16%, was similar to that of a rural watershed. Based on 95% CI overlap, the percentages of P_{Qwp} estimates equal to zero were not significantly different between rural and transition watersheds, but both were greater than that for suburban-urban watersheds (Table 3-6; Figure 3-8). White Perch of a size of interest to anglers were more likely to be found in subestuaries with rural or transition watersheds and least likely to be found in subestuaries with suburban-urban subestuaries.

The level of development in Mattawoman Creek's watershed more than doubled between 1989 (0.43 C/ha) and 2011 (0.91 C/ha; Figure 3-9). This watershed

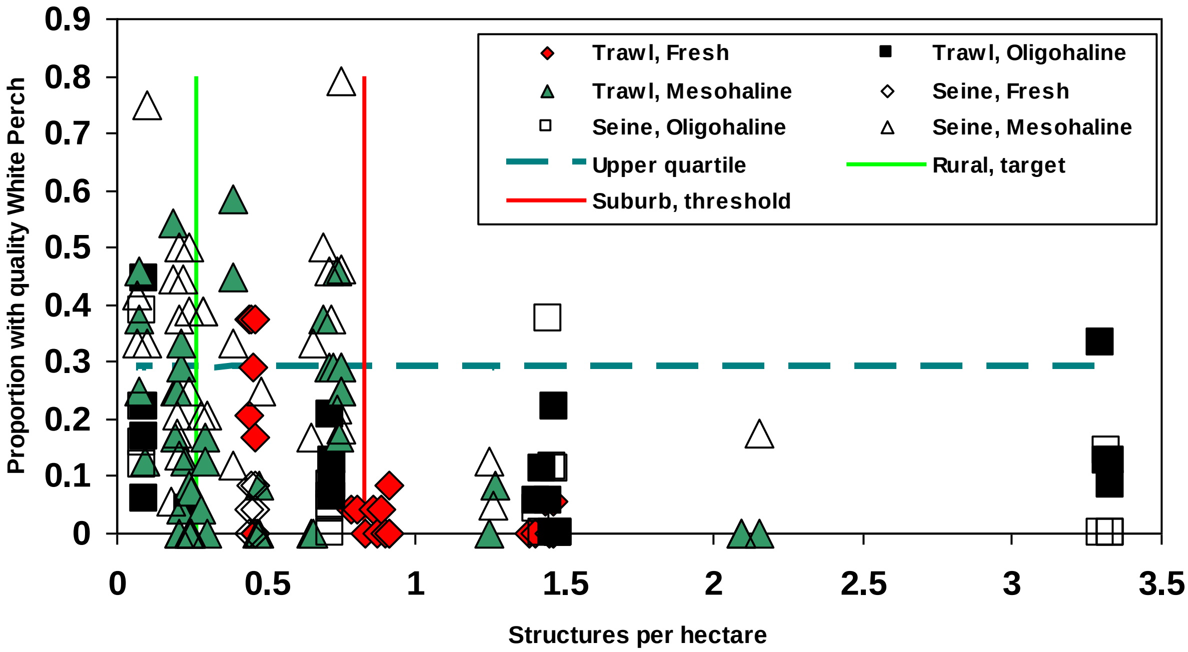

Figure 3-8. Proportion of annual subestuary samples with quality-size (≥ 200 mm) White Perch (P_{Qwp}) during 2002013, by gear (seine or trawl) and salinity class. Upper quartile of P_{Qwp} is indicated by dotted horizontal line and lower quartile equals 0. Vertical green line indicates boundary for rural watershed target and vertical green line indicates boundary for suburban watershed threshold.

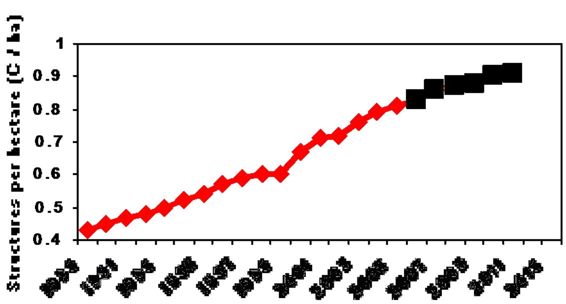

Figure 3-9 . Trend in development (structures per hectare or C / ha) of Mattawoman Creek's watershed during 1989-2013. Black square indicates values that are at or beyond the threshold for a suburban watershed.

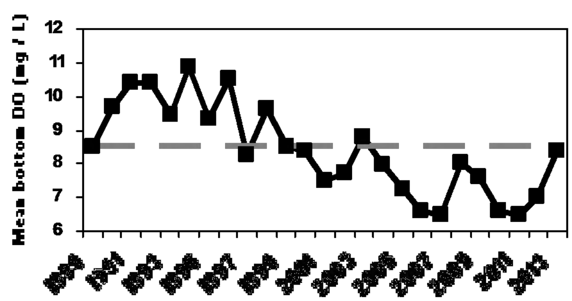

Figure 3-10. Mean bottom dissolved oxygen (DO) during July-September in Mattawoman Creek's subestuary, 1989-2013. Dotted line indicates median for the time-series of annual means.

reached the threshold for suburban development (C/ha = 0.83) in 2006 (Figure 3-9).

There appeared to be two periods of bottom DO in the Mattawoman Creek time-series (Figure 3-10). Mean bottom DO was near or above the median for the time-series (8.5 mg/L) during 1989-2000 (C/ha ≤ 0.67) and then fell below the median afterward (with the exceptions of 2003 and 2013). Mean bottom DO in 2013, 8.4 mg/L, was very near the median for the time-series. Annual mean bottom DO has never fallen below the target of 5.0 mg/L and excursions below this level were rare (Figure 3-10). These shifts in bottom DO corresponded to changes in Mattawoman Creek's subestuary chlorophyll a from high (16-40 µg / L) to low (< 15 µg / L) and shift in SAV acreage from low (coverage of ~10% or less of water area) to high (coverage of > 30%; Figure 3-11; Uphoff et al. 2011; 2012).

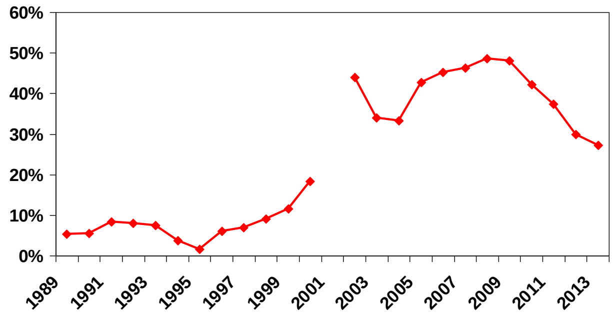

Figure 3-11. Percent of Mattawoman Creek's subestuary covered by SAV during 1989-2013.

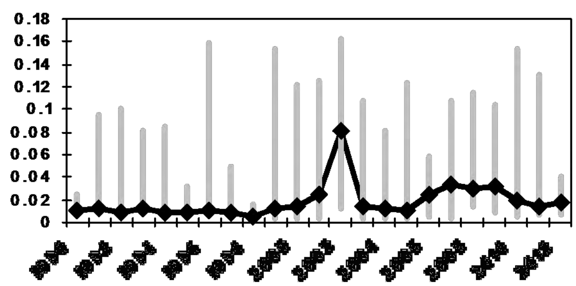

Figure 3-12. Range (vertical line) and median mg/L of total ammonia nitrogen (TAN) at a Chesapeake Bay Program monitoring station in Mattawoman Creek during SAV growing season.

Total ammonia nitrogen exhibited two time periods corresponding to those observed for bottom DO (Figure 3-10) and SAV (Figure 3-11). Median TAN was low and stable at 0.01 mg/L or lower through 2000 and then began a rapid rise to a spike of 0.08 mg/l in 2002 (Figure 3-12). Median TAN dropped after 2002, but was elevated beyond that seen prior to 2001; during 2007-2009 median TAN was consistently elevated at 0.03 mg/L. Estimates of median TAN were generally much closer to minimum than maximum estimates. Maximum estimates of TAN were 2-6 times higher than their respective medians, while differences between the minimum and median were much less (Figure 3-12).

Geometric mean catches and their 95% CIs for 3.1 m and 4.9 m trawls are presented in Table 3-5. The linear regression of GM catches of 4.9 m and 3.1 m trawls during 2009-2013 were closely and linearly related (r^2 = 0.93, α = 0.008, N = 5). The slope was significant (α = 0.0079), but the intercept was not (α = 0.58) and we predicted missing portion of the 3.1 m trawl GM time-series from the slope alone (Figure 3-13). The span of GMs in the regression was similar to

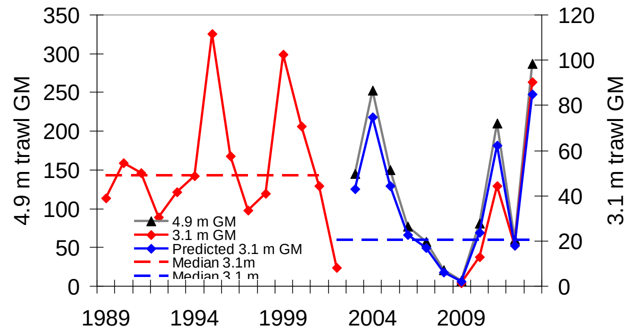

Figure 3-13. Geometric mean (GM) catches per trawl of all species of finfish in Mattawoman Creek during 1989-2013. Note dual axes for 3.1 m and 4.9 m trawls. Predicted 3.1 m GM is based on a linear regression of 3.1 m and 4.9 m trawl GMs during 2009-2013. Dotted horizontal lines indicate median GM of 3.1 m trawl samples for 1989-2001 (red dotted line) and 2002-2013 (blue dotted line).

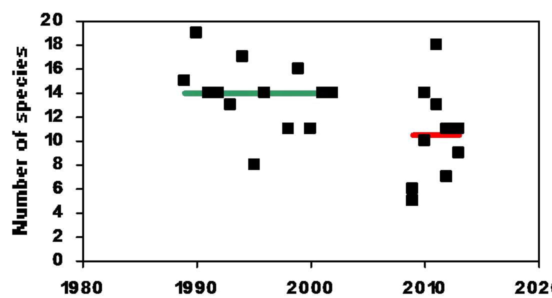

Figure 3-14. Species richness (number of species) in 3.1 m trawl samples during summer sampling. N = 12 for all points. Bimonthly sampling during 2009-2013 allowed for two estimates of N = 12 per year. Median number of species during 1989-2002 is indicated by the green line; median number of species during 2009-2013 is indicated by the red line.

those that were predicted, so values did not have to be extrapolated. The full 3.1 m GM time-series (observations and predictions) suggested a downward shift in total abundance in 2002. During 1989-2001, minimum, maximum, and median GM catches of all species were 30.3, 111.7, and 48.7, respectively; during 2002-2013, minimum, maximum, and median GM catches of all species (predictions for missing years included) were 1.5, 90.2, and 20.3, respectively (Figure 3-13).

Mattawoman Creek's finfish abundance appeared to be susceptible to boom and bust dynamics after 2001. The "busts" were concurrent with spikes (2002) or plateaus (2007-2009) of TAN. Collapses of the magnitude exhibited during 2002 and 2008-2009 were not detected previously (Figure 3-13). Uphoff et al. (2010) determined that the collapse of abundance in 2008-2009 was local to Mattawoman Creek and not widespread in the Potomac River.

Species richness in 3.1 m trawl samples declined between 1989-2002 and 2009-2013 (Figure 3-14). During 1989-2002, minimum, maximum, and median number of species collected annually were 8, 19, and 14 respectively; during 2009-2013, minimum, maximum, and median annual number of species collected annually were 5, 18, and 10.5, respectively (Figure 3-14). Between 1989-2002 and 2009-2012, Uphoff et al. (2013) found that the proportion of 3.1 m trawls with Bluegill had increased noticeably; Spottail Shiner, Tesselated Darter, Striped Bass juveniles, and Pumpkinseed were largely unchanged; presence of White Perch declined noticeably; and planktivorous Blueback Herring, Alewife, Gizzard Shad, and Bay Anchovy declined drastically.

Shifts in ecosystem status such as that observed in Mattawoman Creek may represent shifts to different unstable or stable states (shifting baselines or regime shifts, respectively) of ecological systems rather than steady declines (Steele and Henderson 1984; Duarte et al. 2009). The term "regime shift" has been used to suggest jumps between alternative equilibrium states are nonlinear, causally connected, and linked to other changes in an ecosystem (Steele 1996; Duarte et al. 2009). The regime shift concept implies that different regimes have inherent stability, so that significant forcing is required to flip the system into alternative states (Steele 1996). Eutrophication is one of these forcing mechanisms (Duarte et al. 2009), while urbanization creates a set of stream conditions (urban stream syndrome; Hughes et al. 2014a; 2014b) that qualifies as a shift as well. Both of these processes (eutrophication and urban stream syndrome) are interrelated products of development in Mattawoman Creek's watershed. Sediment loads in Mattawoman Creek from construction and stream bank erosion were high (Gellis et al. 2008) and increased nutrient loading there was strongly associated with sediment level increases that occurred after 2003 (J. Uphoff, MDDNR, unpublished analysis of USGS data obtained by W. Romano, MDDNR). Approaching and breaching the development threshold in Mattawoman Creek's watershed has been concurrent with changes in stream hydrology and water quality, increased sediment and nutrient loading from stream erosion and construction, decreased chlorophyll a (a powerful indicator of ecosystem response to nutrients; Duarte et al. 2009) and DO, increased water clarity, TAN and SAV, and decreased finfish abundance (particularly planktivores) and diversity in the subestuary (Gellis et al. 2008; Uphoff et al 2009; 2010; 2011; 2012; 2013). These changes have been persistent for over a decade.

When evaluated in the context of Chesapeake Bay Program's habitat goals, Mattawoman Creek superficially resembles a restored system with reduced nutrient loads, i.e., increased clarity, reduced chlorophyll a, and increased SAV. Together, these factors were expected to increase habitat for fish (Chesapeake Bay Program 2014). However, Chanat et al, (2102) reported that nutrient and sediment loads in Mattawoman Creek were nearly twice those of the Choptank River, an agriculturally dominated watershed twice the size of Mattawoman Creek. Boyton et al (2012) modeled nutrient inputs and outputs in Mattawoman Creek and found that nutrients were not exported out of the subestuary, suggesting that wetlands, emergent vegetation, and SAV in Mattawoman Creek were efficiently metabolizing and sequestering nutrients. Unfortunately, the response of the fish community has not been positive. Duarte et al. (2009) analyzed responses of phytoplankton of four coastal ecosystems to eutrophication and oligotrophication and found diverse and idiosyncratic responses. An expectation that ecosystems would revert to an expected reference condition was unsupported (Duarte et al. 2009). The overall declines in finfish abundance and diversity in spite of improved clarity and SAV exhibited in Mattawoman Creek may indicate that achieving these goals of the Chesapeake Bay Program may not lead to improved fish habitat in some subestuaries.

Finally, here we develop a hypothesis that water quality dynamics in Mattawoman Creek's extensive SAV beds (low DO, high pH, and high organic matter) may be creating episodes of ammonia toxicity for fish. Ammonia is considered one of the most important pollutants in the aquatic environment because it is both common and highly toxic (US EPA 2013). Ammonia toxicity in fish is heavily influenced by pH; temperature and salinity are considered minor influences (Randall and Tsui 2002). Low DO may lead to positive feedback of nutrient cycling, enhancing NH_4 levels (Testa and Kemp 2012). The toxic substance profile for ammonia developed by the United Kingdom's Marine Special Areas of Conservation Project (2001) determined that toxicity of ammonia increased with low DO.

Breakdown of organic matter is a source of ammonia (US EPA 2013). Macrophyte beds have high primary productivity and are an important source of organic matter (Caraco and Cole 2002). The microorganisms of decay assimilate some of the organic material in the dead remains to build their cells (Cole 1975). Other organic material is converted to ammonia. This, in turn, is oxidized to nitrite and then to nitrate. Both aerobic and anaerobic bacteria function in ammonification, while only aerobic forms participate in nitrification (Cole 1975). Low DO in SAV beds can impact this biogeochemical cycle within the bed (Caraco and Cole 2002).

Some species of SAV create low DO conditions and introduced species, in particular, may induce hypoxia (Caraco and Cole 2002). Uphoff et al. (2011) found low DO patches were not uncommon within an extensive SAV bed in Mattawoman Creek and DO conditions were generally worse within the SAV bed than in bottom channel waters. Introduced Hydrilla and Eurasian Milfoil are commonly encountered in Mattawoman Creek and often form dense beds (J. Uphoff, MD DNR, personal observation). In general, SAV are two orders of magnitude less sensitive to ammonia than aquatic invertebrates and vertebrates (US EPA 2013).

Toxicity of ammonia to fish increases with pH (Randall and Tsui 2002; US EPA 2013) and conditions within SAV beds are in a range where enhanced toxicity could be expected. Growing season (April-October) median pH during 2002013, measured at the continuous monitor within the Sweden Point Marina SAV bed from Maryland DNR's Eyes on the Bay (http://mddnr.chesapeakebay.net/eyesonthebay/), ranged between 7.7 and 8.2, while maximum pH varied from 8.9 to 9.6.

If toxic ammonia caused episodic "disappearances" of Mattawoman Creek's estuarine fish community, it either did so at levels well below EPA's acute criteria for aquatic life (US EPA 2013) or at levels much greater than indicated by TAN monitoring at MAT0016. Under the temperature and pH conditions used by US EPA (2013) for chronic ammonia conditions (longer term reductions in survival, growth, or reproduction), the range of TAN maximum measurements at MAT0016 (0.08-0.015 mg/L) and the Sweden Point continuous monitor measurements of pH and temperature indicate a potential match for pH from 8.6 to 9.0 at water temperatures from 21 to 30 ºC during 2002011. Measurements of TAN from the Chesapeake Bay Program's monitoring site MAT0016, while adjacent to the continuous monitor at Sweden Point Marina, are channel measurements. These measurements may be diluted by mainstem Potomac River tidal inflow. Anecdotally, we have observed multiple fish kills in Mattawoman Creek since the early 2000s. Some have followed tournament releases of Largemouth Bass at Sweden Point Marina; at least one was fairly widespread and involved multiple species.

Randall and Tsui (2002) criticized ammonia criteria for fish because they were based on starved, resting, non-stressed fish. These criteria did not protect swimming and stressed fish, nor did criteria reflect that feeding reduced the toxicity response. Fish may employ strategies, such as reduced ammonia excretion, that ameliorate ammonia toxicity (Randall and Tsui 2002).

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Job 2: Support multi-agency efforts to assess and delineate interjurisdictional finfish habitat and ecosystems

Jim Uphoff, Margaret McGinty, Alexis Park, Carrie Hoover, Bruce Pyle, Paul Parzynski

Introduction

The objective of Job 3 was to document participation of the Fisheries Habitat and Ecosystem Program (FHEP) in habitat, multispecies, and ecosystem-based management approaches important to recreationally important finfish in Maryland's Chesapeake Bay and Atlantic coast. Contributions to various research and management forums by Program staff through data collection and compilation, analysis, and expertise are vital if Maryland is to successfully develop an ecosystem approach to fisheries management.

Maryland Fisheries Service – Fisheries Habitat and Ecosystem Program Website

We continued to populate the website with new reports and information to keep it up to date with project developments. We are working on a new website design that will launch in 2014, which will be easier to navigate and contain additional valuable information, such as an *Angler's Toolkit*.

Publications

Blazer, V. S., and coauthors (includes J. Uphoff). 2013. Reproductive health of yellow perch *Perca flavescens* in selected tributaries of the Chesapeake Bay. Science of the Total Environment 447:198-209.

Jacobs, J. M., R. M. Harrell, J. Uphoff, H. Townsend, and K. Hartman. 2013. Biological reference points for nutritional status of Chesapeake Bay striped bass. North American Journal of Fisheries Management 33:468-481.

A final manuscript, *Striped Bass and Atlantic Menhaden Predator-Prey Dynamics: Who is Driving the Bus*?, (J. Uphoff and A. Sharov) has been submitted for an AFS book.

Environmental Review Unit Bibliography Database

FHEP staff continue to compile an Environmental Review Unit database, adding recent literature and additional topics including effectiveness of Best Management Practices. We also purchased Endnote Software to house the Bibliography and C. Hoover is in the process of entering the bibliography into Endnote. Bibliographies for the Striped Bass AFS manuscript and Yellow Perch stressors for the State's Yellow Perch workgroup were created, and additional topics on Environmental Site Design, Ecological/Ecosystem Services, Ecosystem Based Management, and Social Science aspects have been added.

DNR Interagency Effort on Mattawoman Creek

M. McGinty continued to support efforts to promote conservation of Mattawoman Creek by attending meetings of citizen groups and local government officials to communicate the ecological value of Mattawoman Creek and recommend planning strategies conducive to conservation. We supplied comments on fisheries for DNR's review of the draft Charles County Comprehensive Plan, dated November, 2012. DNR's comments reflected concerns about inconsistency between many of the key provisions within the County's preferred draft plan document and the legal requirements and intent of Maryland law for planning and natural resource conservation. Under the county's preferred plan, development would be sprawl-based and would increase well beyond the threshold for productive fisheries in Mattawoman Creek and Port Tobacco River. Charles County did not adopt this plan and their political process continues to churn towards a final comprehensive growth plan.

DNR Habitat Matrix Team

M. McGinty participated in the agency's habitat matrix team, providing support in addressing development projects in Maryland that threaten to alter terrestrial and aquatic habitat. This group is also developing outreach and communications materials outlining the value of natural landscapes.

Database Development

Scientific Collection Permits (SCP) are issued by the State to groups (agencies, organizations, individuals) who wish to legally try to collect finfish, shellfish, other target species, or data in the State of Maryland waters. They in return submit a report on their findings providing the location, date, species collected, number count, and gear used, or any other parameters collected.

Through the auspices of the State of Maryland, the permit coordinator authorizes the validity of the requests based on standard parameters consisting of location, time of year, gear used, type of species targeted and number collected, and use of such data. Other restrictions may be applied based on newly updated regulations.

These findings from the collectors are then scrutinied to identify which data would be appropriate to include in a database. Data from 2003 to present has been archived and the invaluable information is now being prioritized and recorded.

After ascertaining whether the data is from tidal or nontidal waters, the tidal waters finfish data was entered by B. Pyle on an Excel worksheet. It consists of permit number, location (coordinates if available), scientific (genus/species) as well as common name of fish, number of fish, collecting agency and any pertinent comments.

The SCP data from year 2003 to present will be used in conjunction with MBSS's nontidal data for future map plotting and verifications.

P. Parzynski is using ArcGIS 10.1 to build a GIS database using water quality and fish presence/absence data from the SCP data, VIMS's ChesMMap and NEAMAP programs, and the Chesapeake Bay Program's data library. The creation of these geospatial relationships will help FHEP better indentify critical fish habitat. These maps are intended to shed light on main bay juvenile habitat to compliment FHEP's data on spawning habitat.

Corsica River Restoration Support

M. McGinty, A. Park, and C. Hoover provided the Corsica Implementers a summary of the tidal fish community from 2004-2011 in the Corsica River for the 2012 Corsica River Public Report titled *Corsica River Targeted Initiative: Progress Report and Watershed Plan Addendum 2005-2011*. Upon the presentation of the summary, no improvements or declines in the tidal fish community were indicated for the Corsica River; however, staff recommended a conservative approach to development based on the extensive monitoring and research that had been conducted since 2003.

Cooperative Research

A. Park, C. Hoover, and P. Parzynski supported field sampling efforts of various state and federal projects including: the DNR's Coastal Bays Program, Resident Species Program, Fish Passage, the Alosid Project, and Smithsonian Environmental Research Center (SERC) herring monitoring and nearshore habitat projects. J. Uphoff collaborated with NOAA, University of Maryland, and West Virginia University scientists at Oxford Laboratory on the development of nutritional reference points for Striped Bass.

We, along with additional MD DNR programs, worked to identify key metrics to be included in the Bay Report Card. Staff provided indicators of development with a brief description of land-use impacts on fish habitat. This will include development thresholds and what they mean in a fisheries management context.

Presentations and Outreach

The following technical presentations were given during the project year.

More Pavement Equals Less Fish: Stream Spawning of Anadromous Herring Declines with Development at the 27th Annual AFS Tidewater Chapter conference.

Organic Matter Matters to Yellow Perch Larvae in Chesapeake Bay Subestuaries: Watershed Development Impacts Early Feeding Success at the annual meeting of the American Fisheries Society.

Managing Land Use, Fish Habitat and Estuarine Fisheries in a Developing Watershed at the Chesapeake Bay Program's Designing Sustainable Coastal Habitats Workshop.

Managing Land Use, Fish Habitat and Estuarine Fisheries in a Large, Diverse Watershed at the Partnership for the Delaware Estuary 2013 annual meeting plenary session.

Managing Land Use, Fish Habitat and Fisheries for DNR's Sportfish Advisory Committee.

How's the buffet? Nutrition and Striped Bass, part 2 (part 1 was by NOAA's John Jacobs) of a tandem presentation on Striped Bass nutrition reference points in Chesapeake Bay for the Chesapeake Bay Program's Fisheries Goal Implementation Team.

Nutritional Targets and Limits for Chesapeake Bay Striped Bass for the ASMFC Biological and Ecological Reference Point Committee.

Developing Management Strategies to Conserve High Priority Fisheries Habitat for the Oyster Advisory Commission; the Conservation Education Division Retreat and the Environmental Review Division.

C. Hoover and P. Parzynski organized and led sampling and fish identification training at the 16th Annual Bush River Wade in. C. Hoover presented sampling results and led volunteer training for the Anita C. Leight Estuary Center staff and volunteers. The Bush River is one of our sampling areas. This volunteer group samples the Bush River and provides data to the project.

C. Hoover participated in TEAM training, through which staff will be certified to assist with outreach activities. TEAM is dedicated to educating elementary and middle school students about the Chesapeake Bay and other natural resource issues in Maryland by helping students understand and care for their natural environment.

A. Park, C. Hoover, and P. Parzynski participated in various outreach events to demonstrate seining techniques and familiarize volunteers, students, and the public with common fish species of the Chesapeake Bay. Events included: Annual BioBlitz at Anita C. Leight Estuary Center, Fishing Week at Easton YMCA, and Fish Identification Training at Jug Bay

M. McGinty, A. Park, C. Hoover, and P. Parzynski participated in four separate teacher's training events throughout the state of Maryland in October for Governor O'Malleys' Explore and Restore Your Schoolshed Teacher Development, to demonstrate biological, chemical, and physical sampling that teachers in coastal counties can do in relation to the Stream Restoration work.

M. McGinty, A. Park, C. Hoover, and P. Parzynski attended the Maryland Water Monitoring Annual Conference and presented in the Healthy Watersheds Session. The presentation communicated results associating development with fisheries losses, impacts of contaminants associated with developed lands and the need to conserve productive landscapes to assure sustainable fisheries for the future. Citizen scientists described the impact of development on spawning habitat in Mattawoman Creek. The presentation included data provided by FHEP staff.

J. Uphoff, A. Park, C. Hoover, P. Parzynski, and J. Thomspon created a small video documenting sampling methods and spoke on the decline of water quality due to increasing development for the public. The film will be posted on the Fisheries Habitat and Ecosystem Program's website in 2014.

M. McGinty organized and participated in a session held at the Alliance for the Chesapeake Bay's Annual Watershed Forum. The session presented the science supporting the linkages between development and fish habitat and how we are applying the science to management with a presentation, *The Science: Impervious Surface Impacts on Tidal Fish.*

M. McGinty presented the impervious surface work to the Port Tobacco Watershed Society to local citizens interested in becoming actively involved in Fisheries Habitat and Ecosystem Program's monitoring studies.

J. Uphoff, A. Park, C. Hoover, and P. Parzynski participated in a field interview with Tom Pelton from WYPR radio over the issues of watershed development and fish habitat.

Staff Development

We participated in a day long workshop discussing the Chesapeake Bay River Herring at Smithsonian Environmental Research Center (SERC). The workshop is part of a new project at SERC to develop methods for run counts of river herring in the Chesapeake Bay using imaging sonar. The goals of the workshop are to provide fishery researchers an overview of SERC's proposed work, to exchange information on research activities, findings, and objectives, to identify shared research goals, and to develop collaborative relationships to achieve those goals.

We attended the "Water Words that Work" workshop held by the Maryland Water Monitoring Council, learning how to promote environmental issues and science to the public. This training was beneficial and taught staff how to effectively communicate your environmental message with the general public.

M. McGinty, A. Park, and C. Hoover attended the Watershed Resources Registry (WRR) Technical Action Committee (TAC) workshop held by the Office of Sustainable Futures. This training helped provide insight on GreenPrint and other interactive development tools.

M. McGinty, A. Park, and C. Hoover attended the Baltimore Washington Partners for Forest Stewardship presentation titled *Forest Ecosystem Services: Valuing Nature's Benefits to People and Our Local Communities*. The presentation helped staff gain knowledge in communicating with and teaching local citizens how they can help keep water clean.

ASMFC

J. Uphoff provided an update of the status of Weakfish was provided to the Weakfish Technical Committee and Board. Non-age structured indicators of weakfish status were updated through 2012, including exploitable biomass and juvenile indices, Proportional Stock Density length quality indices, relative F and relative exploitation. A run was made with a predator-prey biomass dynamic model to estimate biomass, F, and M for judging relative status of weakfish. Weakfish biomass remained very low, slightly better than 2010-2011, but still ranking among the lowest measured. Landings and estimated discards rose from 123 MT in 2011 to 311 MT in 2012. Fishing mortality rose from a very low point in 2011, but is still among the lowest measured. The stock is exhibiting little response to substantial reductions in F. Proportional Stock Densities indicate that very few weakfish are reaching harvestable size. A run of the biomass dynamic model with a predation/competition term indicates that M is still extremely high (1.08 in 2012) compared to F (0.13).

J. Uphoff was appointed to the Biological and Ecological Reference Point Committee. Major activities were reported above and in Job 4.

Environmental Review Support for Estuarine and Marine Habitat

B*ob Sadzinski*

Environmental review and planning represents the "frontline" of habitat management. The direct link between land-use, ecological condition of downstream receiving water and environmental review provides the opportunity to mitigate the impacts of land-based projects on aquatic resources through the permitting process.

The Task Force for Fishery Management recognized that Maryland Department of Natural Resources' (DNR's) Environmental Review (ER) Program was critically understaffed (Task Force on Fisheries Management 2008). An Integrated ER Team was created by assigning personnel from various units throughout DNR to address this critical staffing shortfall. Fisheries Service has provided one reviewer and an advisor who provides additional expertise to project review topics as well as guidance in setting environmental review policy for the Department. The activities of these positions are funded through this federal aid grant: ER activities were entirely funded under Job 2, while advisory and support activities were also covered under Jobs 1 and 3.

The Environmental Review unit has been charged by the Secretary of Natural Resources with both conducting routine reviews and taking a lead role in proactively using habitat criteria in project review activities. Routine reviews may be streamlined by developing habitat criteria for triage, such as impervious surface reference points and greater application of GIS technology.

The purpose of environmental review is to work proactively with partners (other DNR agencies, Maryland's Department of Environment and Department of Planning, local governments, and federal agencies) to protect key habitats and ecosystem functions and limit environmental impacts while making better natural resource data available to agencies at the state, county and local levels. Environmental review must identify the natural resources potentially impacted, assess the extent of the impacts on resources, review for regulatory requirements, and as applicable, identify and attempt conflict resolutions. The review agency is responsible for providing comments based on potential

impacts of the project on the resources of concern to that agency and recommends avoiding, minimizing or mitigating project impacts as appropriate.

Major Activities in 2013

In 2010, DNR had assigned two staff members as the primary environmental reviewer and planner (Bob Sadzinski) and the other as the liaison for the Fisheries Service (Jim Uphoff), but in 2011, Bob Sadzinski became both the reviewer and the Fisheries Service liaison. In 2013, this position was also assigned four more counties: Harford, Cecil, Kent and Queen Anne's.

For the environmental reviewer and planner, duties included estuarine and marine environmental reviews for Charles, St. Mary's, Calvert, Prince George's, Anne Arundel, Cecil, Harford, Kent and Queen Anne's Counties and all statewide landfill, reef and aquaculture applications. Table 1 presents an overview of the number of projects by permit type. In 2013, 438 permit applications were reviewed, many of which required significant DNR coordination.

	Number of Projects Reviewed			
Application Type	2011	2011[a]	2012	2013[b]
Aquaculture	24	14[c]	7	11
Reef	1	4	2	2
Living Shoreline	NA	64	36	25
County-Specific	141	250	296	398
Surface Mine	10	16	4	0
Landfill	18	14	6	2
Total	194	362	351	438

Table 1. Overview of the projects by application type and year. ([a]Two additional counties were assigned to the reviewer in 2001; [b]four additional counties were assigned to the reviewer in 2013; cThe environmental Review Unit ceased reviewing aquaculture permits in 2011, The process was streamlined in 2011, however, occasional requests come from the Army Corps of Engineers.

In addition, the environmental reviewer/planner served as an advisor for programs including Smart Growth, Green Infrastructure, Blue Infrastructure, BayStat/StateStat, and Plan Maryland. We cooperated and coordinated the various landscape-based DNR habitat initiatives and utilized information developed by these programs. These programs were responsible for providing multi-disciplinary information to key partners;

- Codifying regulatory standards for water quality, especially for the key quantitative parameters that define limits of acceptable habitat quality for important species
- Identifying and prioritizing high quality aquatic habitats for protection, and

Developing key stream management strategies and comprehensible living shorelines, climate change and comprehensive plan policies.

One of the most significant project developments was the increase in county responsibilities which resulted in significant increase in annual permit reviews. This ensured improved coordination for tidal projects since these counties all had tidal areas and improved coordination with Fisheries Service to identify and protect essential fish habitat including shallow water habitat.

Potential future projects include developing a framework to enhance sound coastal and marine resource conservation, management and restoration by:

- Completing detailed spatial assessments of coastal habitat, critical natural resources, and associated human uses

- Identification and prioritization of areas containing concentrations of sensitive aquatic habitats and resources including essential fish habitat.

Continue to restructure the current GIS system to include additional pertinent data layers including aquatic bottom types and navigational channels.

Job 3: Developing Priority Fisheries Habitat Spatial Tools

Margaret McGinty, Rachel Uphoff, Paul Parzynski, Bruce Pyle

One of the top priorities identified by the Task Force on Fisheries Management (2008) was developing strategic, quantitative habitat criteria to promote better management of fish habitat. This is becoming a greater priority as habitat investigations in Maryland are showing losses related to land use change. (Uphoff et al 2011a; 2011b; 2012; 2013). Population in Maryland is projected to increase by another 1.1 million people by 2030 (Maryland Department of Planning 2011), with an attendant increase in urbanization to accommodate this growth. Increased development has been identified as a threat to the health and recovery of the Chesapeake Bay (Chesapeake Bay Program). Uphoff et al. (2011a) documented the impact of increased impervious surface on fish habitat and populations, proposing impervious surface targets and limits. Investigation of the relationships of fisheries productivity to development-related reference points has continued under Federal Aid to Sportfishing Grant F-63 (Uphoff et al. 2011; 2012; 2013). As development impacts a greater portion of a watershed, effectiveness of fisheries management on resident species shifts from harvest control to landscape management, habitat conservation, rehabilitation, and reengineering (Uphoff et al. 2011a). In the Chesapeake Bay region, many of these responsibilities now lie with agencies not involved in fisheries management. Fisheries managers need to effectively and openly communicate potential loss of quality of life, sustainability, and services (fish, fishing opportunities, and ecological services) due to degraded habitat so that stakeholders, responsible agencies, and governing bodies can make informed, overt decisions about trade-offs between development and conservation of rural landscapes needed for fisheries (Uphoff 2011a).

We are applying development targets and thresholds in spatial tools to help local planners and land managers promote sound planning that conserves key fish habitats as Maryland accommodates this projected growth. We are in the process of identifying and mapping habitat based on target species occupation. We are applying management priorities related to habitat quality to help target appropriate fisheries and landscape management approaches. Based on definitions of restoration terminology, we applied the terms conservation, rehabilitation and re-engineering (Society of Ecological Restoration, 2004), to define land management strategies that best describe realistic management expectations for fisheries. In our experience, many shifts in aquatic habitat with development are nonlinear "flips" into persistent negative states (shifting baselines or regime shifts, respectively) rather than steady declines (Steele and Henderson 1984; Duarte et al. 2009). Shifting baseline and regime shift concepts imply that once a negative "flip" has occurred due to development, restoring them to an idealized past reference state by reducing human-induced pressures proportional to their past increases is unlikely (Steele 1996; Duarte 2009). The goal in rehabilitation is to repair ecosystem processes, productivity and services (SER, 2004). We believe rehabilitation is a more feasible goal in this context, because even in areas moving from rural to early suburban, development permanently alters a watershed (see Job 1, sections 1-3 for examples), precluding a complete return to pre-disturbance conditions. We promote re-engineering (ecological engineering) in highly disturbed urban watersheds, defined as "manipulation of natural materials, living organisms and the physical-chemical environment to achieve specific human goals and solve technical problems" (SER 2004). We suggest this is a feasible management recommendation since the watershed has been highly altered from its natural state and is now dominated by engineered features.

This job describes progress to date in developing criteria to delineate and depict critical fisheries habitat. Several examples of the tool's application are provided.

General Approach: We are developing habitat criteria for all life stages of target species that occur in Maryland tidal waters using historical and recent target species presence and-or abundance data. We begin by identifying natural limiting factors to distribution of each species and life stage. Where we have data, we develop cumulative frequency distributions of presence and or abundance by each limiting factor. Salinity influences distribution and abundance of fish (Hopkins and Cech 2003; Cyrus and Blaber 1992; Allen 1982), and we interpret it as the dominant natural factor shaping distribution of fish in the Bay. We are developing salinity criteria for each target species and life stage occurring in Maryland's portion of the Chesapeake Bay, by season of occurrence (Table 1). In the absence of distribution by salinity data, we use values from the literature to define natural limits. We use cumulative frequency of presence and-or abundance by salinity of a species and life stage to categorize habitat as preferred (high occurrence), acceptable (modest occurrence), marginal (low occurrence) and not suitable (absence). Cumulative distributions generally exhibit four stanzas of change with salinity that we translate into habitat classes. Preferred habitat is indicated over the points with the most rapid change in frequency of occurrence with salinity. As changes

Species	Life History Classification	Life Stage	Season of Occurrence By Life Stage
Alewife Herring	Anadromous	Spawning	Spring
American Shad	Anadromous	Spawning Juvenile	Spring Summer
Blueback Herring	Anadromous	Spawning	Spring
Striped Bass	Anadromous	Spawning Juvenile Adult	Spring Summer Year Round
White Perch	Resident Semi-Anadromous	Spawning Juvenile Adult	Spring Summer Year Round
Yellow Perch	Resident Semi-Anadromous	Spawning Juvenile Adult	Spring Summer Year Round
Atlantic Menhaden*	Marine Migrant	Juvenile	Spring- Fall
Spot*	Marine Migrant	Juvenile	Spring to Fall
Gizzard Shad	Freshwater Forage	Spawning Juvenile Adult	Year Round Year Round Year Round
Slivery Minnow	Freshwater Forage	Spawning Juvenile Adult	Year Round Year Round Year Round
Spottail Shiner	Freshwater Forage	Spawning Juvenile Adult	Year Round Year Round Year Round

Table 1. Target species life history and seasonal occurrence of life stage in the Chesapeake Bay. *Note: Summer distribution data for Spot and Atlantic Menhaden represent distribution of adult and juvenile life stages combined. These species were not identified by life stage in the database.

in cumulative frequency slow, the next stanza of modest change characterizes acceptable habitat. Beyond this and before a species/life stage is absent, is a range of salinity associated with very slow change in frequency signifying marginal habitat. We use straight lines that best fit the portions of the cumulative distribution to define habitat categories.

We use interpolated average seasonal bottom salinity data obtained from the Chesapeake Bay Program data to map these areas in the Bay (Tom Parham, Resource Assessment Service, personal communication). Figure 1 shows average bottom salinity in the spring in Maryland's tidal waters of the Bay. Maps were produced using ArcMap 10 from Environmental Systems Research Institute (ESRI 2011). We scored habitat by category, assigning a score of 5 to preferred habitat, 3 to acceptable, 1 to marginal and 0 to no occurrence. This score represents the habitat score for a given area of the Bay.

Once natural distributions were identified, we de-

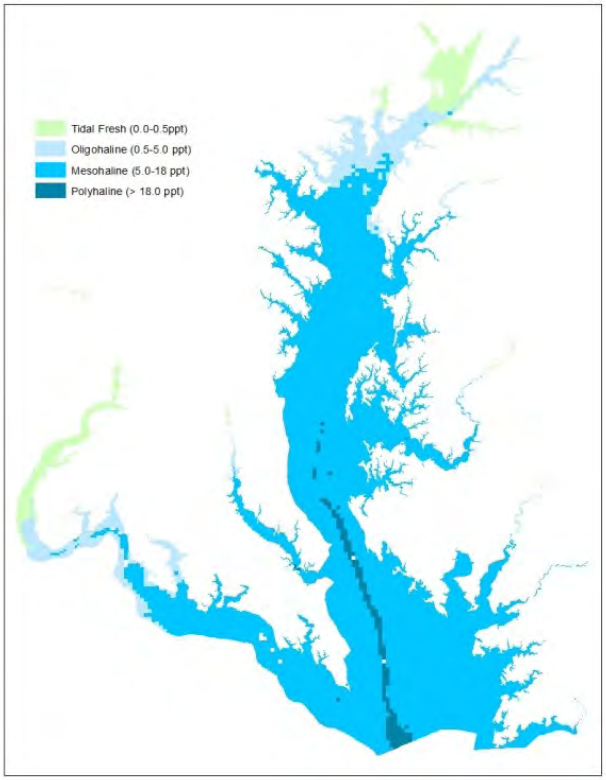

Figure 1. Spring salinity by zone in the Chesapeake Bay.

Figure 2. Impervious surface targets and limits in the tidal watersheds of Maryland.

fined stressors that impact natural distribution and map those. Stresses of urbanization to fish have been the focus of our study. We have applied our targets and thresholds by watersheds in Maryland using Towson University Impervious Surface Data (Figure 2). We scored watersheds based on the percentage of impervious cover level, where watersheds with impervious surface less than 5% (target impervious level) received the highest score of 5; watersheds with impervious surface between 5 and 10%, a score of 3; and

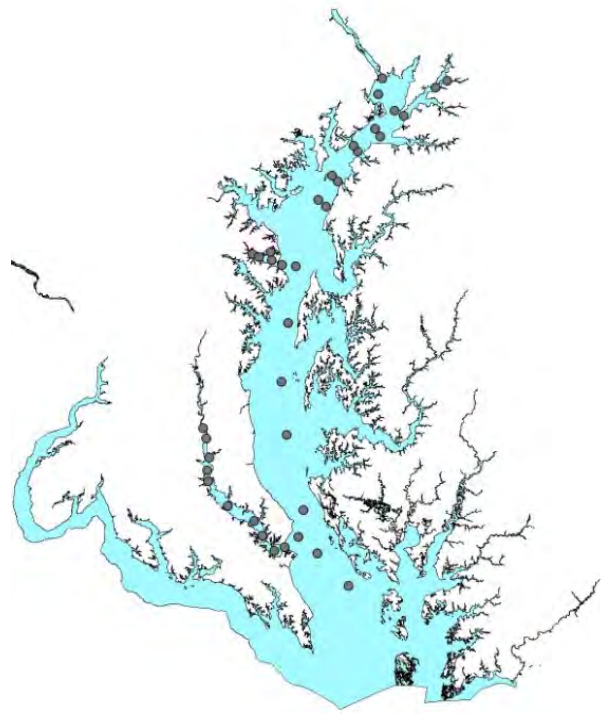

Figure 3. Approximate egg sampling locations in the Patuxent River and Upper Chesapeake Bay, Maryland (Dovel, 1971).

watersheds with impervious surface between 10 and 15%, a score of 1. Watersheds exceeding the 15% threshold received a rank score of 0.

We combined habitat data with the watershed impervious surface data and summed the habitat and watershed scores to derive a total habitat value score between 0 and 10. We ranked the total score into terciles and assign the upper tercile a score of 5, the middle a 3 and the lower a 1. If the species score was 0, then the total score was 0 because these areas are not potential habitat for the species life stage. This avoids assigning a low habitat value score to an area that would not naturally support the species' life stage. We mapped watersheds based on the total habitat value

Species	Preferred	Acceptable	Marginal	No occurrence
Alewife spawning	0-1 (99.2%)	1-2 (0.6%)	2-3 (0.2%)	>3
American Shad spawning	0-1 (100%)			
Blueback Herring spawning	0-1 (99.5%)	1-2 (0.5%)		>2
Striped Bass spawning	0-3 (99%)	3-9 (1%)		>9
White Perch spawning	0-1 (99%)	1-2 (0.4%)	2-10 (0.6%)	>10
Yellow Perch spawning	0-2*			

Table 2. Categorized salinity ranges for anadromous and semi-anadromous spawning habitat in Maryland tidal waters of the Chesapeake Bay. Percentages represent the percentage of observations in each category.

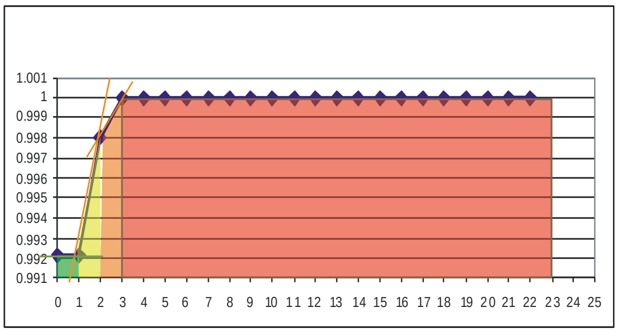

Figure 4. Cumulative frequency of occurrence by salinity for Alewife. Lines were fit to determine cut-offs. Green areas indicate preferred habitat (0-1 ppt), yellow, acceptable (1-3ppt) and orange, marginal (3-11 ppt). There were no occurrences of alosid eggs or larvae in salinities greater than 11 ppt as indicated by the red shaded area.

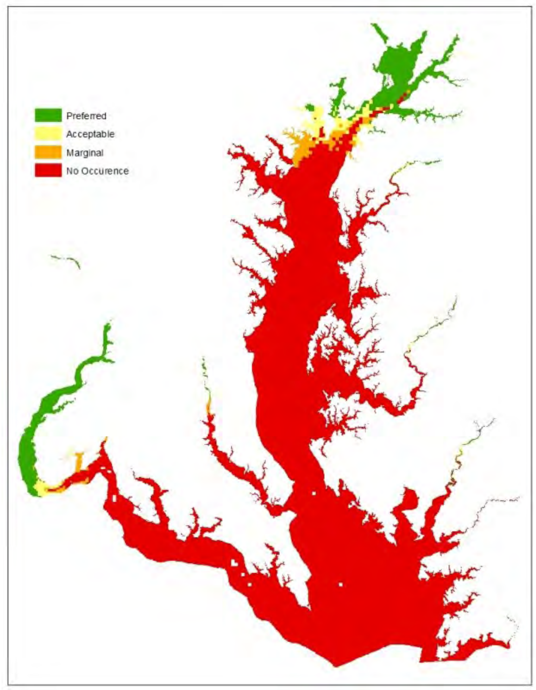

Figure 5. Alewife spawning habitat by category.

score and assign management priorities based on these ranks. Watersheds that scored a 5 were designated as high priority habitat conservation areas for fisheries because their ecological functions related to fisheries were considered intact. Areas with a rank of 3 were designated as fisheries habitat rehabilitation areas. Watersheds with a rank of 1 are identified as habitat areas in need of reengineering. Watersheds scoring 0 are not assigned a management priority, because they do not represent habitat for the given species and life stage.

Anadromous and Semi-Anadromous Spawning Habitat: We defined salinity preferences for anadromous and semi-andromous spawning habitat for target species, including Alewife, American Shad, Blueback Herring, Striped Bass, White Perch and Yellow Perch based on egg surveys in the upper reaches of the Chesapeake Bay and the Patuxent River conducted between 1960 and 1968 (Dovel 1971; Figure 3). Abundance of eggs for each species by salinity was used to calculate cumulative frequency by salinity. (Larval catch data was combined with juvenile catch data, so spawning habitat was designated by egg abundance alone.) We fit lines to the distributions to determine salinity categories by life stage and species. Habitat categories for the anadromous species and their life stages are presented in Table 2. Figure 4 shows an example for Alewife where lines are fit to designate categories.

We applied salinity criteria to develop maps characterizing spawning habitat in Maryland's tidal waters for anadromous and semi-anadromous spawning habitats in Maryland. These maps represent the natural salinity limits to distribution of spawning in Maryland's Chespeake Bay. Figure 5 is an example for Alewife, showing the geographical extent of habitat

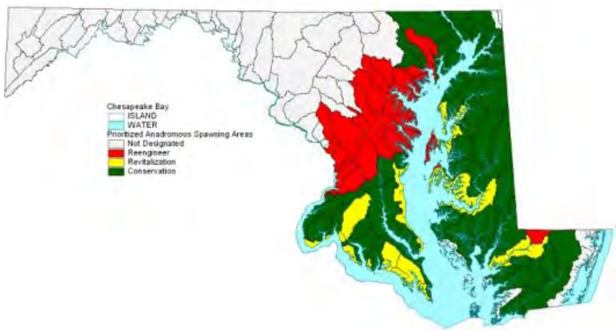

Figure 6. Anadromous and Semi-anadromous spawning habitat designations in Maryland, by management actions.

by category.

We scored the categories according to our ranking approach and combined these ranks with watershed ranks. We applied this same approach to all anadromous species combined and developed a map that identifies management priorities for anadromous and semi-anadromous spawning habitat in Maryland (Figure 6).

Juvenile Target Species Habitat - We obtained historical (1959 to present) summer salinity and juvenile catch data for Maryland's portion of Chesapeake Bay from the Juvenile Striped Bass Seine Survey (Durell and Weedon 2014). Numerous stations were sampled during that time frame in rivers known to support Striped Bass spawning (Durell and Weedon 2014). We also obtained historical seine data (1989-1992) from sampling that was previously conducted in numerous smaller tributaries to the Chesapeake Bay, for development of an Index of Biological Index (Carmichael et al, 1992), using the same seine survey techniques as the Striped Bass Seine Survey. We combined these data with seine data collected in our summer Estuarine Fish Sampling (Job 1, Section 4), which also applied the same methodology used in the Striped Bass Seine Survey. Figure 7 shows the sampling sites covered in these three combined studies. We compiled these data into one dataset since methods were identi-

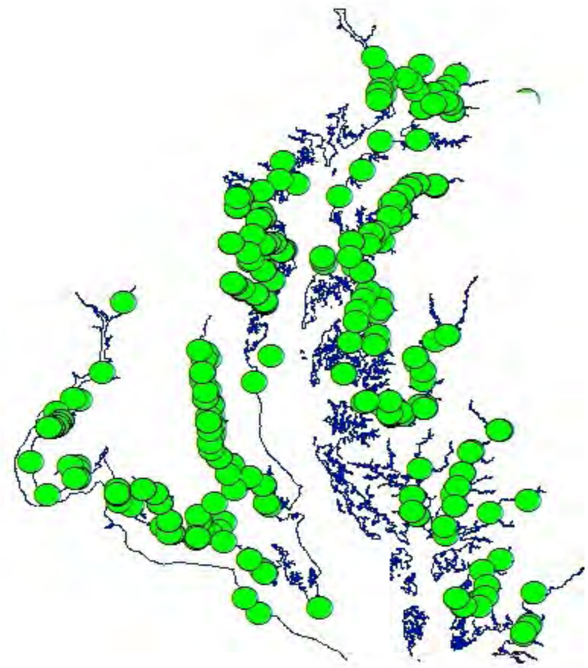

Figure 7. Seine sties sampled in all three surveys combined.

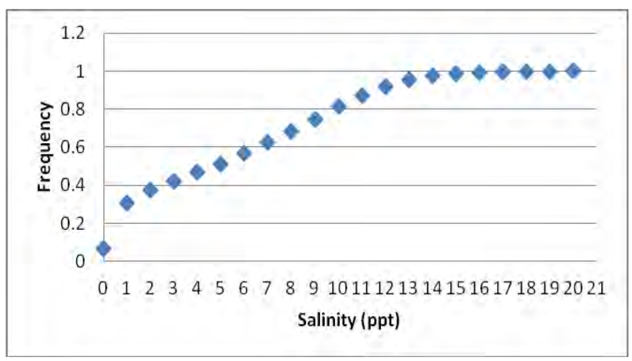

Figure 8. Frequency of samples by salinity at 1 ppt increments.

cal, to develop salinity criteria and categories for juveniles and unclassified life stages of target species. These surveys specify juvenile life stages for anadromous and semi-anadromous species, but group juveniles and adults together for other target species.

Salinity in these surveys ranged from 0 - 21‰, however, 98% of the samples were collected in salinity ranging from 0 - 14 ‰ (Figure 8). We mapped average bottom salinity for the summer in Maryland's Chesapeake Bay to determine how much of the area of the Bay was represented by the data. In general, summer salinity in all areas (except for the deep channel)

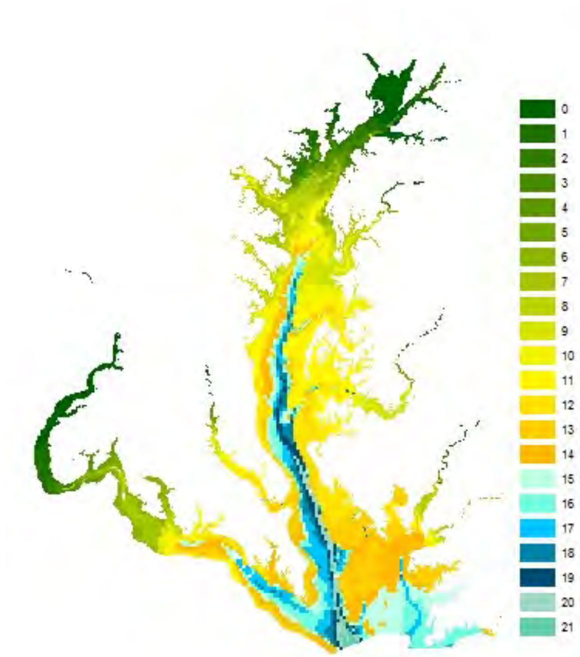

Figure 9. Mean summer bottom salinity in the Maryland portion of Chesapeake Bay.

averaged between 0 and 14 ‰ (Figure 9).

We calculated proportion of samples with species (S*p*) present by life stage for each 1 ppt increment

between 0 and 20 ppt. Because sampling effort varied with salinity, we divided this proportion by effort (n) to adjust for sampling effort (adjusted S*p*= S*p*/n*)*, summed the adjusted Sp estimates and recalculated the cumulative percentages by salinity from effort-adjusted estimates to classify habitat for each species. We applied the same general approach described previously to evaluate habitat categories for juvenile life stages. Table 3 shows the salinity criteria by species and life stage and the percentage of observations by each category for juveniles of target anadromous and semi-anadromous spawners. Table 4 shows salinity criteria for marine target species. The criteria for marine target species represents criteria for juvenile and adult life stages combined, because catch data for these species did not identify life stage. We will apply the same approach to these data to define salinity criteria for target freshwaters species.

We are exploring other approaches to verify these criteria. We are testing an abundance metric that may be more sensitive for schooling species such as Atlantic Menhaden, where frequency of occurrence is low, but abundance is high. We will compare abundance

Species	Preferred	Acceptable	Marginal	No occurrence
Alewife	0-5 (64%)	5-10 (22%)	10-19 (16%)	19-20
American Shad	0-1 (44%)	1-4 (37%)	14 (19%)	120
Blueback Herring	0-3 (52%)	3-8 (30%)	8-19 (18%)	19-20
Striped Bass	0-14 (80%)	120(20%)		
White Perch	0-6 (59%)	6-13 (32%)	13-19 (9%)	19-20
Yellow Perch	0-5 (76%)	5-10 (22%)	10-14 (2%)	120

Table 3. Salinity limitations for anadromous and semi-anadromous juvenile life stage by

Species	Preferred	Acceptable	Marginal	No occurrence
Atlantic Menhaden	1-9 (59%)	9-17 (30%)	0-1, 17-20 (11%)	
Spot	5-19 (84%)	0-4 (16%)		19-20
Atlantic Croaker	2-13 (75%)	13-15 (14%)	0-2; 15-18 (11%)	18-20

Table 4. Salinity limitations for marine species juvenile and adult stages combined. Percentages represent the percentage of observations in each category.

metrics to presence metrics to determine which better describe distribution and habitat occupation. Once completed, we will evaluate stressors and proceed in developing maps that reflect management priorities for juvenile habitat in the Bay.

We will also seek to obtain data to develop criteria for the adult life stages of those target species Maryland. We have mapped historical fishing spots in Maryland. We are also developing a spatial data base from scientific collection permit data. These data along with trawl surveys may provide suitable data to achieve this goal.

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JOB 4: Development of ecosystem-based reference points for recreationally important Chesapeake Bay fishes of special concern: Striped Bass nutrition and forage availability benchmarks

Jim Uphoff, Jim Price (Chesapeake Bay Ecological Foundation), Bruce Pyle, and Carrie Hoover

Abstract

We evaluated linkages of a proposed nutritional target for Striped Bass with average weight or calories of prey eaten, and forage availability in Maryland's portion of Chesapeake Bay during October-November, 2006-2012. Attainment of target nutritional status (low vulnerability to starvation) was indicated when 30% or less of Striped Bass were without body fat. Most sublegal and legal Striped Bass sampled were vulnerable to starvation. Chances of reaching the target were less than 1% for legal fish in four of seven years and six of seven years for sublegal fish. In remaining years, there was a 44-100% chance that fish met the target. Nutritional state of sublegal fish was closely related to grams of prey consumed per gram of Striped Bass during October-November, but nutritional state of legal fish was not. Although five major prey items were identified, both grams and calories of prey eaten by both size classes of Striped Bass were usually dominated by Atlantic Menhaden even though their relative abundance was low.

Introduction

Reports of Striped Bass in poor condition and exhibiting ulcerative lesions increased in Chesapeake Bay during the mid-to-late 1990s (Overton et al. 2003; Gauthier et al. 2008), spurring concerns about the effect of low Atlantic Menhaden abundance on Striped Bass health (Uphoff 2003). Uphoff (2003) determined that these phenomena could be linked with poor feeding success on Atlantic Menhaden *Brevortia tyrannus*. Mycobacteriosis emerged in Chesapeake Bay in the late 1990s and an epizootic has affected Striped Bass in Chesapeake Bay (Jiang et al. 2007; Gauthier et al. 2008; Jacobs et al. 2009); challenge studies have linked nutrition and mycobacteriosis (Jacobs et al. 2009). Tagging and epidemiological models have provided evidence of increased *M* (total annual instantaneous natural mortality rate) of Striped Bass in Chesapeake Bay that is concurrent with the mycobacteriosis outbreak (Jiang et al. 2007; Gauthier et al. 2008; Sadler 2010). High *M* of Chesapeake Bay Striped Bass may have serious implications for management since this stock is the main contributor to Atlantic coast fisheries (Richards and Rago 1999; Sadler 2010).

Uphoff et al. (2009) identified a need for diet sampling and condition or nutritional health indicators for Striped Bass to address concerns about the effect of low forage abundance on Striped Bass *Morone saxatilis* well-being in Chesapeake Bay. Jacobs et al. (2013) evaluated Fulton's condition factor, relative weight, percent moisture, and an index of coverage of viscera by visible body fat (body fat index) as indicators of Striped Bass lipid content (nutritional status) in five experiments. Proximate composition is the standard for judging nutritional condition, but it is expensive. Statistical models developed for both moisture content and the body fat index (including presence or absence of body fat) adequately predicted tissue lipids, offered clear indication of lipid depletion, and would be far less expensive to implement than proximate composition for routine monitoring of nutritional status. Jacobs et al. (2013) reported a threshold for body moisture, but a visible body fat target (< 30% of Striped Bass without fat) was also developed (J. Jacobs, NOAA, personal communication). Fulton's condition factor and relative weight indices were poorly related to Striped Bass lipid concentration in experiments (Jacobs et al. 2013).

Lipids serve as the energy currency in marine fish (Rose and O'Driscoll 2002) and are the source of metabolic energy for growth, reproduction, and swimming (Tocher 2003). Changes between feeding successfully and starving influence lipid allocation among storage, maintenance, and growth (Jacobs et al. 2013). Starvation caused declines in energy reserves, physiological condition, and enzyme activity in Atlantic Cod *Gadus morhua*, degeneration of swimming muscle in Winter Flounder *Pleuronectes americanus*, and increased natural mortality of Atlantic Cod (Dutil and Lambert 2000). Natural mortality may not be immediate and could be delayed after unfavorable conditions (Dutil and Lambert 2000).

Jacobs et al. (2013) stressed that comparisons of body fat to nutritional criteria (the body fat target or moisture threshold) should be based on October-November since the criteria for Chesapeake Bay Striped Bass were developed from samples during that time span. A citizen based predator-prey monitoring effort by Chesapeake Bay Ecological Foundation (CBEF) has collected Striped Bass diet and condition data in mid-Chesapeake Bay (mid-Bay) since 2006 under a Maryland Department of Natural Resources (MD DNR) collector's permit. Tagging has indicated that most Striped Bass that do not join the coastal migration remain within Maryland's portion of Chesapeake Bay and many are found in this mid-Bay region (Cimino and Johnson 2009).

In this federal aid report, we document steps needed to transform raw data from CBEF's paper ledgers into an Excel spreadsheet data base for 2006-2012. We also have included descriptions of winter (December – March) diet data because 2007-2008 collections were provided to the ASMFC Biological and Ecological Reference Point workgroup.

Annual October-November Striped Bass diets were summarized by numerical, weight, and caloric composition. Numbers of prey ingested provide insight into feeding behavior, while weight and caloric content of prey consumed reflect nutritional value (MacDonald and Green 1983; Pope et al. 2001).

We examined linkages of the proportion of Striped Bass without body fat (Pf0) with average number, weight, or calories obtained, and forage availability through graphical, correlation, and regression analyses. Attainment of "safe" nutritional status (low vulnerability to starvation) was judged by comparing Pf0 to a proposed target of 30% or less of Striped Bass with Pf0 (John Jacobs, NOAA, personal communication). We compared CBEF indicators of nutritional

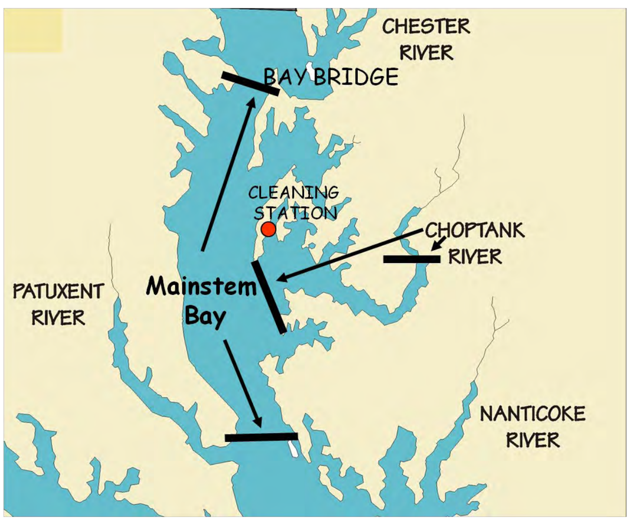

Figure 1. Area of Maryland's portion of Chesapeake Bay sampled during 2006-2012. Bars indicate approximate boundaries of mid-Bay and Choptank River regions. Dot indicates cleaning station where charterboat catches were sampled.

condition (absence of body fat and body fat scores indicating successful feeding) to the same indicators derived from monitoring by MD DNR's Fish and Wildlife Health Program to validate and interpret CBEF monitoring results.

Methods

Field Collections Year-round collections were made voluntarily by James E. Price in a portion of Maryland's mainstem Chesapeake Bay bounded approximately by the William Preston Lane Bay Bridge to the north, the mouth of Patuxent River (excluding Choptank River; hereafter, mainstem Bay) to the south, and the Choptank River from its mouth to Warwick Creek to the east (Figure 1). Collectively, this mainstem Bay region and Choptank River will be referred to as the mid-Bay.

Active trips were made to collect Striped Bass by jigging, casting, trolling, and occasionally by bottom fishing with bait. Conditions of the collectors permit allowed for samples of up to15 sublegal (< 457 mm TL; hereafter sublegal Striped Bass or fish) and 15 legal fish ($\geq$ 457 mm TL; hereafter legal Striped Bass or fish) per trip. A typical trip lasted 2-6 hours and usually occurred in late afternoon or evening, with a few trips extending into night. Most active trips occurred in Choptank River, but some occurred in the mainstem Bay. These trips were the only source of sublegal fish. On many trips, an effort was made to collect Striped Bass from more than one location. In some cases, fish were simply not available except in one location and the sample was drawn from there.

Fish kept as samples during active trips were placed in a cooler in the boat with ice in warm weather or in a cooler without ice in cold weather. Shortly after a trip, fish were either worked up immediately by J. Price or held on ice. Fish held on ice were usually worked up the next day, but might be held for an additional day or two.

Striped Bass were sampled at a Tilghman, Maryland, check station as well (Figure 1). These were legal fish caught by charter boats that chummed, fished chunks of Spot or Atlantic Menhaden, fished live (live-lined) Spot, or trolled. These trips occurred in daylight. Fish were from a mix of morning, afternoon-evening, or all day charters. Striped Bass would have been iced immediately. Fish, minus fillets, were collected over one to several days (depending on how many boats were chartered and how successful they were) by the proprietor of the fish cleaning service, held on ice, and worked up at the check station by J. Price.

Fish Examination, Data Entry, and Editing - Data were recorded in a ledger (usually by Henrietta Price) that contained a year's data. Date of sampling and a location where the fish were caught was recorded. Location was often specific enough that a general location known to fishermen could be determined (False Channel, Stone Rock, a buoy number, etc.), but sometimes only very general locations (Choptank River or Chesapeake Bay) were available. If a fish was caught by gear other than hook-and-line, it was noted in a comment.m This did not occur during October-November sampling, but winter samples could have been obtained from Striped Bass caught in gill nets. Each fish on a sample date was assigned a fish number noting sequence of processing. The total length (TL) of each Striped Bass was recorded to the nearest quarter of an inch. If whole, the fish was weighed to the nearest ounce on a calibrated and tared spring scale. A digital scale was used to measure small Striped Bass and some food items. The body cavity was opened, sex was determined, the spleen was observed and scored for nodules (a possible indicator of mycobacterial infection), and body fat was classified. The classes for extent of nodules ranged in whole numbers from 0 (nodules absent) to 3 (high presence). The body fat index ranged from 0-4 (no body fat to complete coverage of viscera). Body fat between 0 and 1 was assigned a fractional score, while scores from 1-4 were usually whole numbers (with a few half -scores, 1.5 for example). The gut was then opened and contents identified. Contents were classified as whole or partially intact (latter was noted). Total length of intact and partially intact fish and shrimp, carapace width of crabs, and shell length of intact bivalves were measured (usually inches to the nearest quarter, but occasionally mm). Soft and easily digested small items such as amphipods or polychaetes were recorded as present or were assigned a fullness class (these items were largely absent in fall, but were more common in late winter – early summer). Abbreviations indicating status of gonads were recorded as were occasional comments (Table 1).

These ledgers were provided by CBEF to FHEP and entered into an Excel spreadsheet. Each year's data was recorded in a worksheet. Variable names and labels were standardized and additional variables were created (Table 1). In general, the data entry and editing process attempted to standardize variable names and labels, convert English units to metric (lengths in mm and weights in grams), and provide estimated weights of food items and Striped Bass without weights.

A sequence number was assigned to each data line. The sequence was somewhat arbitrary, but provided a way of getting data into original order after manipulations. Location was split into three general areas (variable = "Area") for where Striped Bass were caught: Choptank (River), Bay (mainstem Chesapeake Bay), and Ocean (sampled in winter). If a more specific location was noted, it was recorded under "Location". A column was created for total lengths filled into each data line ("TL_mm fill") to provide an opportunity to examine prey to predator length ratios. Another column with a single TL for each Striped Bass sampled ("TL_mm") was created. Stomach contents, as they were labeled in the ledgers, were recorded in "Stomach Contents" and then a standard label was assigned to each type of item under "Contents edit". Partially intact diet items were denoted by a "1" under "Part_1"; a blank in this column indicated an intact item. Food item TL, in inches, was recorded under "Food_in", converted to TL in mm under "Food_L_mm", and then converted to grams (described below) under "Food wt". In some years, gizzards of Atlantic Menhaden or Gizzard Shad (latter were much less frequently encountered) were measured and these measurements were recorded under "Gizzard_mm"; gizzards are relatively hard and persistent in the guts of Striped Bass. Striped Bass weights in English units were divided into separate columns for pounds ("Lb.") and ounces ("Oz."), and then were converted into grams ("gms"). Striped Bass that were measured, but not weighed were assigned an estimated weight ("Estimated gms"; described below; Table 1). Weight of a Striped Bass that was weighed was entered in "Estimated gms".

October-November length-weight regressions ($\log_e$-transformed lengths and weights) from Choptank River Striped Bass samples were used to estimate missing weights (mostly mainstem Bay fish). The equation for these estimates was

$$\log_e(W) = a + b \cdot (\log_e L);$$

where W was weight in grams, L was total length in mm, a was the intercept, and b was the slope (Pope and Kruse 2007). The estimate of $\log_e(W)$ was exponentiated to estimate weight.

Diet items were identified to species, when possible, or assigned to the lowest taxonomic group identifiable. Table 2 provides the label assigned under "Contents edit name", the standard common name or taxonomic category, genus and species epitaph, diet status (an indicator of use in analyses; 0 = not used and 1 = used), and whether the item was found in the Chesapeake Bay or Ocean (0 = absent and 1 = present). Two parameters for a non-linear allometry equation for converting diet item length to weight (grams; Hartman and Brandt 1995c) are provided in Table 2. In a few cases, equations for a similar species were substituted when an equation was not available. These diet item allometry equations were used to reconstruct diets for Overton (2003; 2009), Griffin (2001), and Griffin and Margraf 2003), and were originally developed by Hartman and Brandt (1995c). Allometry equations described changes in diet item weight (W) with length as

$$W = a \cdot (L^b);$$

where lengths (L) were TL for fish and shrimp, carapace width (CW) for crabs, or shell length (BL) for bivalves; a is a constant and b is an exponent (usually between 2.5 and 4.0 for fish; Pope and Kruse 2007; Table 2).

Data Analysis – Two groups of Striped Bass were formed for analysis of October–November diet: suble-

Entered variable	Description	Edited variable	Description
location	Specific or general	Sequence	Number assigned to each line for year
year		Date	Derived from month day for each year
month		Area	Choptank R., Bay, or Ocean
day		location	Specific when available
gear	Hook and line unless noted	year	
fish #	Sequence of fish examined on a day	month	
TL(in)	Total length of Striped Bass in inches	day	
Sex	Male, female, unknown	fish_num	Sequence of fish examined on a day
Stomach Contents	Diet item label; may not be standardized	TL_in	Total length of Striped Bass in inches
Partial (1)	Blank = intact; 1 = part identifiable to species	TL_mm fill	TL in mm assigned to each line of data
inches	Length of diet item, inches	TL_mm	TL in mm assigned once to Striped Bass
mm	Length of diet item, mm	Sex	Male, female, unknown
Spleen	Nodule classification; 0-3; 0 = no nodules	Stomach Contents	Diet item label; may not be standardized
B. Fat	Classes 0-4; subclasses < 1	Contents edit	Standardized diet item label
Lb.	Striped Bass weight measured, pounds	Part_1	Blank = intact; 1 = part identifiable to species
Oz.	Striped Bass weight measured, remaining	Food_L_in	Length of diet item, inches
gms	Striped Bass weight measured, grams	Food_L_mm	Length of diet item, mm
Gonad	Abbreviation indicating status	Food wt	Estimated weight of diet item, grams
Comments	Miscellaneous	Gizzard_mm	Size of fish gizzard
		Spleen	Nodule classification; 0-3; 0 = none
		B. Fat	Classes 0-4; subclasses < 1
		Lb.	Striped Bass weight measured, pounds
		Oz.	Striped Bass weight measured, remaining ounces
		gms	Striped Bass weight measured, grams
		Estimated gms	Striped Bass weight (grams) estimated from TL by TL-grams

Table 1. Comparison of variables entered from Chesapeake Bay Ecological Foundation ledgers (Entered variable) and variables created through entry and editing by the Fish Habitat and Ecosystems Program (Edited variable). Descriptions are provided for variables that were considered not to be self-evident.

gal (286-456 mm TL) and legal (457-864 mm TL; hereafter, all lengths are TL unless otherwise noted). The lower limit of the sublegal category, 286 mm was the minimum length in common among years during 2006-2012. Smaller Striped Bass were sampled in some, but not all, years. The upper limit of 864 mm was used to minimize the impact of larger, migratory Striped Bass that reenter the Bay in late fall on estimation of resident diets. This size range was intended to represent resident Striped Bass that did not join the coastal migration after spawning (Cimino and Johnson 2009). These categories accounted for ontogenic changes in Striped Bass diet, but also reflected sample availability (sublegal fish could only be collected by fishing for them directly).

We confined analysis of food items to those we be-

Contents edit name	Common name	Genus species	Diet status	Bay	Ocean	a	b	Comment	Hartman
Amphipod	Gammarus sp.	Gammarus sp.	1	1	0			Not a fall item	
Anchovy	Bay Anchovy	*Anchoa mitchilli*	1	1	1	0.0000005	3.57		Y
Atlantic Herring	Atlantic Herring	*Clupea harengus*	1	0	1	0.0000007	3.6		Y
Bait, chum, chunks, etc			0	1	0				
Blue Crab	Blue Crab	*Callinectes sapidus*	1	1	1	0.0000959	2.86		Y
Blueback Herring	Blueback Herring	*Alosa aestivalis*	1	1	0	0.0000046	3.52		Y
Butterfish	Butterfish	*Peprilus triachanthus*	1	1	0	0.000016	3.08		N[a]
Clam shell			0	1	0				
Clam snout			1	1	0			Small value	
Croaker	Atlantic Croaker	*Micropogonius undula-tus*	1	1	1	0.0000022	3.33		Y
Eel	American Eel	*Anguilla rostrata*	1	0	1	0.000002	2.59	Not a fall item	N[b]
Flounder	Flounder sp.		1	0	1	0.0000056	3.1	Summer Flounder	N[c]
Gizzard shad	Gizzard shad	*Dorosoma cepedianum*	1	1	0	0.0000007	3.6		Y
Goby	Naked Goby	*Gobiosoma bosc*	1	1	0	0.0002088	2.24		Y
Grass shrimp	Grass shrimp	*Palaemonetes pugio*	1	1	0	0.0000047	3.2		Y
Grasshopper			0		0				
Herring	Clupeid		1	1	0	0.0000007	3.6		Y
Mantis Shrimp	Mantis Shrimp	*Squilla empusa*	1	1	1	0.0000047	2.86		Y
Menhaden	Atlantic Menhaden	*Brevoortia tyrannus*	1	1	1	0.0000022	3.35		Y
Mud Crab		*Panopeous*	1	1	0	0.0000959	2.86	Blue Crab	
Mussel	Ribbed mussel		1	1	0			Missed	
None	None		0	1	1				
Oyster shell			0	1	0				
Parasitic arthropod	Isopod		0	1	0				
Pipefish	Northern Pipefish	*Sygnathus fucus*	1	1	0	0.0000007	3.6		Y
Polychaete	Polychaete		1	1	1			Not in fall	
Razor Clam	Razor Clam	*Perkinsus chesapeaki*	1	1	0			Missed	
Regurgitated empty			0	1	0				
Sand shrimp	Grey Sand Shrimp	*Crangon septimspinosa*	1	1	1	0.0000047	3.2		Y
Shrimp	Grass or Sand		1	1	1	0.0000047	3.2		Y
Silverside	Silverside	*Menidia sp*	1	1	1				

Table 2. Summary of information on Striped Bass diet items identified during fall and winter, 2006-2012. Diet status = 1 indicates inclusion in diet estimates and 0 indicates exclusion. Bay = 1 indicates item was identified in mid-Bay during October-November and 0 indicates absence. Ocean = 1 indicates item was identified in Ocean samples during December-March and 0 indicates absence. Parameters a and b are for the allometric length-weight (mm and grams) equations ($Wt = a \cdot L^b$) for items. Under Comments, "Missed" indicates weights not included by mistake. A "Y" under Hartman indicates allometric equation in Hartman and Brandt (1995c) or provided by A. Overton (East Carolina University, personal communication); a "N" indicates an alternative source was used ([a]NEFSC 2004; [b] Unknown; [c] Gilbert 1986; [d] Bradbury et al. 2005).

Contents edit name	Common name	Genus species	Diet status	Bay	Ocean	a	b	Comment	Hartman
Skilletfish	Skilletfish	*Gobiesox strumosus*	1	1	0	0.0000046	3.52	Oyster Toadfish	Y
Soft Clam	Soft Clam	*Mya arenaria*	0	1	0	0.0002341	2.899	Missed	N[d]
Soft invertebrate residue			1	1	0				
Spine (mspine, etc)			0	1	1				
Spot	Spot	*Leistomus xanthurus*	1	1	1	0.0000074	3.13		Y
Tunicate			1	1	0			Mean weight 0.5 gm	
Unknown Crabs			1	0	1			Blue Crab	Y
Unknown fish			1	1	1	0.0000007	3.6		Y
Unknown fish parts			0	1	1				
Unknown residue			0	1	0				
White Perch	White Perch	*Morone americana*	1	1	1	0.0000074	2.95		Y

Table 2. Continued.

lieved were recently consumed in an attempt to keep odds of detection as even as possible. Items with "flesh" (whole or partial fish and invertebrates; crab carapaces were included) were considered recently consumed, while hard, indigestible parts such as gizzards, mollusk shells, and backbones were excluded (Table 2). "Unknown fish parts" were excluded as well (Table 2). Partially intact items with flesh that were identified to species or other taxonomic group were assigned the mean weight estimated for intact items in the same group. Guts classified as "Regurgitated, empty", or with "Unknown residue" were also classified as "none" under "Contents edit name".

Bait was excluded from diet analyses. Whole Spot or White Perch were checked carefully for hook wounds around the head, mid-dorsal, or tail when encountered in samples since they were often used as live bait by charter boats in mid-Bay. Similarly, chunks or pieces of Atlantic Menhaden, Spot, and soft or peeler Blue Crabs might have represented bait or chum (dispersed in water as attractant) and were identified from hook marks or straight knife cuts. Any item identified as bait was omitted from Striped Bass with other diet contents or was classified as "none" under "Contents edit name" if no other items were present.

Feeding metrics were calculated for both subgroups of Striped Bass for each year: proportion of food represented by an item in numbers (PNi), proportion of food represented by an item in weight (PWi), proportion of Striped Bass without food (P_{none}), mean number of an item consumed per Striped Bass (MNi), mean grams of an item per gram Striped Bass (MWi), and mean calories per gram of Striped Bass (MCi; Table 3). Estimates of P_{Ni} and P_{Wi} were based on Striped Bass with stomach contents only, while remaining estimators were derived from all fish sampled including those without food (Pope et al. 2001). Weight of a Striped Bass was represented by measured weights when available or from weight predicted from the relevant length-weight regression when measured weights were absent. Estimates of caloric content of food items (per gram of item; Ci) were from Table 3 of Hartman and Brandt (1995a); estimates at day 305 were applied to October-November data. Estimates of MNi and MWi could be summed to estimate combined annual averages for all items (∑MNi and ∑MWi). Once these metrics were available, a subset of items that accounted for 95% or more of diet by number or weight were identified as major items. Estimates of ∑MCi were based on major items (caloric content estimates of some minor items were not readily available).

Major items were classified as young-of-year or age 1+ based on published size cut-offs or clear modes that could be assigned to age 0 prey. Lengths of major whole items were regressed against the lengths of Striped Bass that ate them to estimate trends in size consumption.

Maryland DNR's Fish and Wildlife Health Program (or FWHP) has monitored Striped Bass health in Chesapeake Bay during fall (late September – November) since 1998 (M. Matsche, MD DNR, personal communication). A categorical body fat index was used by FWHP to evaluate visible reserves of visceral

Metric	Abbreviation	Formula
Proportion without food	P_{none}	Count “None” / count all Striped Bass
Proportion of number of items consumed represented by item i	Pni	Count of item i / Sum of all item counts; fish with food only
Proportion of all grams consumed represented by item i	Pwi	∑ Grams of item i / ∑ Grams of all items; fish with food only
Number of item i consumed per Striped Bass	MN_i	Count of item i / count of all Striped Bass
Grams of item i consumed per gram Striped Bass	MW_i	∑ Grams of item i / ∑ grams of all Striped Bass
Calories of item i consumed per gram Striped Bass	MC_i	$(\sum (MW_i \cdot Ci)) / \sum$ grams of all Striped Bass; C_i = calories per gram of item

Table 3. Feeding metrics, their abbreviations, and formulas that were used to summarize annual Striped Bass diets during October-November, 2006-2012.

body fat: 0 = no detectable fat; 1 = fat present, but coverage was less than 25%; 2 = 25-75% of viscera covered; and 3 = 75% or greater coverage of viscera. Jacobs et al. (2013) analyzed an identical classification to develop nutritional reference points for Chesapeake Bay Striped Bass. These body fat index data, collected by FWHP, were provided to us for analysis with our data by M. Matsche and K. Rosemary.

For both CBEF and FWHP body fat data, the nutrition threshold for individual Striped Bass was indicated by a body fat index of 0 (no visible fat) and the proportion of Striped Bass with that score (Pf0) in the size class sample indicated what fraction met the threshold condition and were vulnerable to starvation (Jacobs et al. 2013). Standard deviations and confidence intervals (95% CI) of Pf0 were estimated from the normal distribution approximation of the binomial distribution (Ott 1977). The probability of meeting a body fat target criterion (see below) equaled the cumulative proportion (expressed as a percentage) of the Pf0 distribution function equaling or falling below the target.

A target level of Pf0 of 30% (John Jacobs, NOAA, personal communication) was used to judge whether mid-Bay Striped Bass had fed successfully during October-November. A target for body fat was not developed by Jacobs et al. (2013), but mean tissue lipid of Striped Bass with a body fat index of 0 was identical to that estimated from percent moisture. Jacobs et al. (2013) presented a target for body moisture (25% or less of fish with starved status) that was derived from mean moisture in fall 1990 field collections and variation in moisture from experiments conducted during 1996-2005 (an estimate of variability of 1990 samples was not available). Feeding conditions were considered favorable in 1990 and these samples offered the only opportunity for a reference condition. Variation of tissue lipids estimated from body fat indices was greater than for moisture and the Pf0 target of 30% for body fat accounted for this additional variation plus a buffer that ameliorated potential for misjudging status (John Jacobs, NOAA, personal communication).

Annual proportions of Striped Bass with body fat indices in excess of 1 (Pff) were calculated for each size class and 95% confidence intervals were constructed as described above. This body fat category quantified the proportion of fish in better condition.

Correlation analysis was used to determine if significant ($\alpha \leq 0.05$) associations existed for annual values of Pf0 or Pff between sublegal and legal size classes. That is, we correlated annual estimates of Pf0 for sublegal fish with Pf0 of legal fish for the CBEF data set; annual estimates of Pf0 for sublegal fish with Pf0 of legal fish for the FWHP data set; annual estimates of Pff for sublegal fish with Pff of legal fish for the CBEF data set; and annual estimates of Pff for sublegal fish with Pff of legal fish for the FWHP data set.

We used linear regression to evaluate how well CBEF based estimates of Pf0 or Pff corresponded to FWHP estimates. Examination of plots of CBEF and

FWHP estimates of Pf0 or Pff did not clearly suggest different relationships for sublegal and legal Striped Bass, so we included both size classes in our regression analyses. Similar to the examination of plots of CBEF and FWHP estimates of Pf0, plots of Pff The regression models could generally be described by the equation:

$$X_{FWHP} = a + (b \cdot Y_{CBEF});$$

where X and Y were estimates of Pf0 or Pff, b was the regression slope, and a was the intercept. If estimates of Pf0 and Pff from both sources were similar, we expected slopes to not be significantly different (at α = 0.05) from 1.0 and intercepts to be zero. Standard output of linear regression analyses tested whether slopes and intercepts were different from zero, so we used an additional two-tailed t-test to test whether slopes were different from 1.0 (Dowdy and Wearden 1991). Deviations from a 1:1 relationship would indicate bias of CBEF estimates and the equations could potentially supply a correction.

Linear regressions of Pf0 with ∑MWi or ∑MCi were used to test relationships of average consumption in weight or calories with nutritional condition of sublegal and legal Striped Bass for both CBEF and FWHP estimates. Results of these regressions (r^2 and α) were compared to see how similarly or differently ∑MWi or ∑MCi were related to nutritional state. CBEF collected legal-size Striped Bass from the mainstem Bay and Choptank River and separate estimates of ∑MWi and Pf0 for these two regions were compared to see if and how often interpretation would have been different from regionally pooled estimates.

Ideally, examination of diets could be eliminated if nutritional state could be strongly related to relative abundance of forage. We used geometric mean indices from MD DNR's long-term (1959-2013) seine survey (Durell and Weedon 2013) as indicators of relative abundance of important fish prey and the density of juvenile Blue Crabs in a winter dredge survey (1989-2013; MD DNR 2013) as an indicator of Blue Crab relative abundance. We assumed these indices would reflect relative abundance of major prey species in Maryland's portion of Chesapeake Bay. Correlation analysis was used to explore the associations of prey relative abundance with MNi or MWi for that species. Ranges of prey indices during 2006-2012 were compared to ranges of indices measured over the entire time-series by dividing the maximum index during 2006-2012 by the full time-series maximum; we assumed that minimums would be approximately the same for the full and partial time-series. While geometric means were used for analyses, each index was divided by their respective 1989-2012 mean in order to place them on the same scale for graphs; this time period was common among all surveys examined. These graphs were split between pelagic and benthic prey.

Results

Samples from 555 sublegal and 1,643 legal sized Striped Bass were analyzed for October-November diet composition during 2006-2012 (Table 4). Number examined during October-November of each year ranged from 47 to 118 sublegal fish and 49 to 327 legal fish. Empty stomachs accounted for 16-57% of samples of sublegal fish during 2006-2012 and 37-63% of samples from legal fish. Nine to 22 dates were sampled and 5-14 separate locations were identified (starting in 2007) during each October-November (Table 4). Each year's sampling usually started during October 1-9 and ended during November 20-30 (Table 5); exceptions were Choptank River 2009 (October 4 – November 16), mainstem Bay, 2010 (October 27 – November 26), and mainstem Bay, 2011 (October 1 – 28).

Length-frequency of all Striped Bass sampled during 2006-2012 shifted upward at the 475 mm length bin, reflecting additional samples of legal Striped Bass from the cleaning station that fell within the 25-mm bin increment (Figure 2); this shift did not reflect the mid-Bay's population size distribution. A summary of October-November length-weight regressions based on $\log_e$-transformed lengths and weights used to estimate missing weights are presented in Table 6.

Twenty-four items were identified in Striped Bass diet samples during October-November, 2006-2012 (Table 7). Atlantic Menhaden comprised 73.7% of the combined years diet by weight and 26.2% by number; Spot comprised 11.3% by weight and 11.7% by number; Blue Crab, 7.5% and 21.5%; White Perch, 3.2% and 2.3%; and Bay Anchovy, 1.7%, and 34.8%. In combination, these five items accounted for 97.4% of estimated diet by weight and 96.5% by number. These items, plus Striped Bass, were used for estimates of ∑MCi; Striped Bass were included due to general management interest. Diet weights of bivalves were not estimated by mistake, but this error should not have affected estimates of PWi or ∑MWi appreciably since they were infrequently encountered.

Bay Anchovy usually accounted for highest PNi of sublegal Striped Bass during October-November (Table 8), 2006-2012, and annually accounted for 19-88% of their diet items by number. Atlantic Menhaden (PNi range = 0.8-30.6%), Spot (PNi range = 0-70.7%), and Blue Crab (PNi range = 0.9-32.8%) were often abundant in each year's diet samples, while White Perch and Striped Bass (primarily young-of-year) accounted for a low fraction of the diet, by number (PNi < 2%). Estimates of PNi of remaining items (combined as "other") varied from 0 to 4.0% (Table

Year	N dates	N locations	Bay	Choptank	N examined	P_{none}
			Sublegal			
2006	19		0	1	118	0.57
2007	12	5	1	1	76	0.45
2008	9	8	0	1	29	0.31
2009	13	5	0	1	99	0.25
2010	18	8	0	1	112	0.19
2011	13	7	1	1	74	0.16
2012	9	9	1	1	47	0.57
			Legal			
2006	19		0	1	49	0.53
2007	20	6	1	1	203	0.44
2008	15	13	1	1	207	0.47
2009	17	12	1	1	240	0.37
2010	22	14	1	1	317	0.4
2011	19	12	1	1	327	0.48
2012	11	10	1	1	300	0.63

Table 4. Comparison of annual sampling of diets of sublegal (<457 mm) and legal (≥ 457 mm) Striped Bass. N dates = number of dates sampled; N locations = number of specific locations indentified (blank = not attempted); Bay or Choptank indicates whether a sample was taken in the region (1) or not (0); N examined = number of Striped Bass examined; and P_{none} is the proportion of fish with empty stomachs.

Year	Region	First date	Last date
2006	Choptank	1-Oct	28-Nov
2007	Bay	Oct	29-Nov
2007	Choptank	2-Oct	20-Nov
2008	Bay	Oct	26-Nov
2008	Choptank	Oct	20-Nov
2009	Bay	30-Oct	25-Nov
2009	Choptank	3-Oct	16-Nov
2010	Bay	27-Oct	26-Nov
2010	Choptank	9-Oct	29-Nov
2011	Bay	1-Oct	28-Oct
2011	Choptank	8-Oct	26-Nov
2012	Bay	7-Oct	30-Nov
2012	Choptank	13-Oct	26-Nov

Table 5. Range of dates sampled for each region and year.

8).

By weight, Atlantic Menhaden (PWi range = 22.8-93.7%) and Spot (PWi range = 0-73.7%) were dominant during 2006-2012 in sublegal Striped Bass diets (Table 8). Bay Anchovy and Blue Crab, although numerous in the diet, accounted for lower fractions of diet weight (1.5-30.9% and 0.1-16.7%, respectively). "Other" items could comprise up to 10.8% of diet weight and made up more than 5% of weight in three years (Table 8). Two of those years had low feeding success in general (2007 and 2011; described below), while Silversides (likely the Atlantic Silverside *Menidia menidia*) and YOY Blueback Herring *Alosa aestevalis* (classified as "other") were present more than usual in 2009 diet samples.

Atlantic Menhaden (PNi = 12.6-76.3%), Spot (PNi = 0-524%), Bay Anchovy (PNi = 5.1-32.5%), Blue Crab (PNi = 2.6-60.4%), and White Perch (PNi = 0-30.4%) were all abundant, by number, in the October-November diet of legal fish at times (Table 9). Striped Bass were present in legal fish diets as low fraction of the diet, by number (PNi < 2%) during 2011-2012. Estimates of PNi of remaining items (combined as "other") varied from 0 to 7.1% for legal sized Striped Bass (Table 9).

By weight, Atlantic Menhaden dominated legal Striped Bass diets during October-November, 2006-2012 (PWi = 55.9-94.7%; Table 9). Bay Anchovy, although sometimes numerous in the diet, accounted for low fractions of diet weight (PWi = 0.1-1.5%). Spot, Blue Crab, and White Perch were absent from diet samples during at least one year (not concurrently) and comprised respective maximum PWi's of 31.7%, 21.0%, and 15.6%. Striped Bass comprised 1.7% of diet weight in 2011 and 6.6% in 2012, but were absent in remaining years. "Other" items comprised up to 5.4% of each year's legal sized Striped Bass diet, by

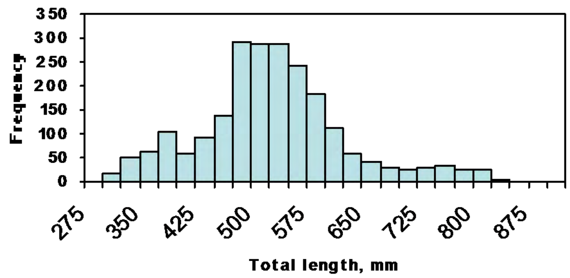

Figure 2. Length-frequency of Striped Bass included in analyses of 2006-2012 diets during October-November.

Year	Slope	Slope SE	Intercept	Intercept SE	r^2	N
2006	3.23	0.06	-13.11	0.36	0.95	167
2007	3.3	0.1	-13.51	0.58	0.96	49
2008	3.18	0.09	-12.62	0.55	0.95	62
2009	3.02	0.04	-11.68	0.22	0.97	199
2010	3.08	0.04	-11.96	0.23	0.96	231
2011	2.89	0.03	-10.96	0.18	0.99	135
2012	3.37	0.07	-13.83	0.44	0.96	95

Table 6. Slopes, intercepts, regression coefficients (r^2), and sample sizes (N) for log_e-transfomed length (mm) versus log_e-transfomed weight of Striped Bass. SE = Standard error.

weight (Table 9).

Nearly all major items were young-of-year, with the exception of White Perch eaten by legal fish. Ninety-seven percent of Atlantic Menhaden eaten by both size classes of Striped Bass (N = 497) were below the age 0 cutoff, as were 96.4% of Bay Anchovy (N = 1,081), 98.0% of Blue Crab (N = 686), and 99.4% of Spot (N = 200). Only 21.1% of White Perch (N = 52) consumed were considered age 0. Size cutoffs for young-of-year were 174 mm TL for Atlantic Menhaden (minimum TL for August-November; ASMFC Atlantic Menhaden Technical Committee, personal communication), 65 mm TL for Bay Anchovy (VIMS 2013), 61 mm CW for Blue Crab (MD DNR 2013), 200mm TL for Spot (VIMS 2013), and 90 mm TL for White Perch (based on the distribution of lengths around the smallest mode).

Lengths of major prey items consumed by sublegal and legal Striped Bass during October-November, 2006-2012, overlapped considerably. Sublegal fish

Diet Item	Item	Wt grams	% weight	% number
Atlantic Menhaden	931	33516.8	73.70%	26.20%
Spot	414	5129.9	11.30%	11.70%
Blue Crab	763	3424.2	7.50%	21.50%
White Perch	83	1457.5	3.20%	2.30%
Bay Anchovy	1235	772.4	1.70%	34.80%
Striped Bass	16	415.1	0.90%	0.50%
Unknown fish	7	224	0.50%	0.20%
Gizzard Shad	5	199.1	0.40%	0.10%
Atlantic Croaker	5	99.1	0.20%	0.10%
Butterfish	4	87.4	0.20%	0.10%
Herring (*Alosa)*	8	42	0.10%	0.20%
Pipefish	2	29.7	0.10%	0.10%
Silverside	11	27.7	0.10%	0.30%
Mantis Shrimp	14	23.6	0.10%	0.40%
Tunicate	26	13	0.00%	0.70%
Mud Crab	9	4.6	0.00%	0.30%
Mumi-chog	1	4.2	0.00%	0.00%
Grass & sand Shrimp	4	1.3	0.00%	0.10%
Goby	2	1	0.00%	0.10%
Amphipod	2	0.2	0.00%	0.10%
Polychaetes	1	0.1	0.00%	0.00%
Clams & razor clams	6		0.00%	0.20%
Mussel	1		0.00%	0.00%

Table 7. Summary of diet items consumed by Striped Bass during October-November, 2006-2012 (combined) by number and weight (grams).

tended to eat smaller sizes and legal fish tended to eat larger sizes of some items. Linear regressions of major prey length against Striped Bass length were significant, positive, and shallow for Bay Anchovy (r^2 = 0.03, P < 0.0001, N = 1,081; Figure 3), Blue Crab (r^2 = 0.08, P < 0.0001, N = 686; Figure 4), and Spot (r^2 = 0.05, P < 0.0001, N = 325; Figure 5), but not for Atlantic Menhaden (r^2 ~ 0.00, P = 0.86, N = 497; Figure 6). The regression for White Perch was not significant at P $\leq$ 0.05 (r^2 = 0.07, P = 0.07, N = 51; Figure 7), but explained as much variation as regressions for Bay

	Sublegal Striped Bass			PNi			
	2006	2007	2008	2009	2010	2011	2012
Total forage Count	85	125	49	490	225	258	35
% Atlantic Menhaden	23.5	0.8	30.6	3.7	7.1	1.9	22.9
% Bay Anchovy	63.5	61.6	46.9	88.0	19.1	66.3	65.7
% Spot	3.5	0	4.1	0.2	70.7	0	0
% Blue Crab	9.4	32.8	18.4	5.3	0.9	28.3	11.4
% White Perch	0	0.8	0	0.2	0.4	0.4	0
% Striped Bass	0	0	0	0	0	1.9	0
% Other	0	4.0	0	2.7	1.8	3.1	0
	Sublegal Striped Bass			PWi			
Total forage grams	699	140	308	1029	1905	314	254
% Atlantic Menhaden	82.7	24.3	88.1	58.9	22.8	33.6	93.7
% Bay Anchovy	2.6	27.9	3.8	26.3	1.5	30.9	5.4
% Spot	13.3	0	5.8	0	73.7	0	0
% Blue Crab	1.5	16.7	2.2	4.4	0.1	12	0.8
% White Perch	0	25.4	0	2.5	0.1	2.3	0
% Striped Bass	0	0	0	0	0	10.5	0
% Other	0	5.7	0	7.9	1.8	10.8	0

Table 8. Annual amounts and percent composition of sublegal Striped Bass diets by number (PNi) and weight (PWi) during 2006-2012.

Anchovy and Spot.

Estimates of ∑MNi for sublegal Striped Bass during October-November varied as much as 6.8-times among years sampled (Figure 8). Estimates were lowest during 2006 and 2012 (∑MNi ~ 0.7 items per Striped Bass), followed by 2007, 2008, and 2010 (∑MNi = 1.6 – 2.0), 2011 (∑MNi = 3.5) and 2012 (∑MNi = 4.9; Figure 8). Bay Anchovies were most numerous in the diet in every year except 2011, when Spot were most numerous (Figure 8). Estimates of ∑MWi for sublegal Striped Bass varied by as much as 6.6-times among years sampled (Figure 9). The lowest estimate occurred in 2007 (∑MWi = 0.004 grams of prey per gram of sublegal Striped Bass), followed by 2011 (∑MWi = 0.007), 2006 and 2012 (0.010), 2008 (0.014), 2009 (0.017), and 2010 (0.025; Figure 9). During years of lowest ∑MWi (2007 and 2011), varying items contributed to the diet of sublegal fish; during remaining years of higher ∑MWi, either Spot (in 2010) or Atlantic Menhaden (remaining years) dominated the diet (Figure 9). Estimates of ∑MCi for sublegal Striped Bass during October November, 2006-2012, varied as much as 7.3-times (Figure 10). The order (lowest to highest) of annual estimates of calories of prey per gram of sublegal Striped Bass during October-November was not different from that indicated by ∑MWi. Differences in relative contribution of prey items to the total diet based on calories or grams consumed were subtle.

Estimates of ∑MNi for legal Striped Bass during October-November, 2006-2012, varied as much as 3.4-times among years sampled (Figure 11). Estimates of ∑MNi of legal fish were lowest during 2012 (∑MNi = 0.5 items per Striped Bass), followed by 2006 (0.8), 2010 (1.2), 2007 (1.3), and 2008, 2009, and 2011 (1.7-1.8). All of the major items were dominant, or nearly so, in ∑MNi estimates during at least one October-November (Figure 11). Estimates of ∑MWi for legal Striped Bass varied by as much as 3.4-times among years sampled (Figure 12). The lowest estimate of ∑MWi occurred in 2012 (0.008 grams of prey per gram of Striped Bass), followed by 2007-2008 and 2010-2011 (∑MWi = 0.010-0.013), 2006 (0.021) and 2009 (0.028). Atlantic Menhaden dominated diet weight of legal fish during October-November (Figure 12). Estimates of ∑MCi for legal Striped Bass during October November, 2006-2012, varied as much as 3.0-times among years sampled (Figure 13). As with sublegal Striped Bass, order (lowest to highest) of annual estimates of calories of prey per gram of legal fish during October-November was not different than indicated by ∑MWi, and differ-

Legal Striped Bass	PNi						
	2006	2007	2008	2009	2010	2011	2012
Total forage Count	38	255	329	438	372	579	161
% Atlantic Menhaden	76.3	33.3	54.7	73.3	26.9	12.6	37.9
% Bay Anchovy	18.4	32.5	26.7	6.8	5.1	23.1	8.7
% Spot	2.6	3.9	5.8	0.9	52.4	0	13
% Blue Crab	2.6	25.1	9.1	10.3	9.4	60.4	5
% White Perch	0	0.4	0.3	1.6	3	1.6	30.4
% Striped Bass	0	0	0	0	0	1.7	1.2
% Other	0	4.7	3.3	7.1	3.2	2.2	5

Legal Striped Bass	PWi						
	2006	2007	2008	2009	2010	2011	2012
Total forage grams	1,140	4,509	3,988	14,702	6,848	5,669	4,405
% Atlantic Menhaden	94.7	83.2	90.6	81.1	62.8	74.6	55.9
% Bay Anchovy	0.3	1.3	1.1	0.1	0.5	1.5	0.3
% Spot	5	8.2	5.3	0	31.7	0	15.5
% Blue Crab	0	3.1	1.3	10.8	2	21	1.2
% White Perch	0	0.6	0.6	3.7	2.5	0.6	15.6
% Striped Bass	0	0	0	0	0	1.7	6.6
% Other	0	3.6	1	4.3	0.6	0.5	5.4

Table 9. Annual amounts and percent composition of legal Striped Bass diets by number (PNi) and weight (PWi) during 2006-2012.

ences in relative contribution of prey items to the total diet based on calories or grams were subtle.

CBEF sample sizes were sufficient for precise estimates of Pf0 during October-November for both size classes of Striped Bass (Table 10). Estimates of Pf0 from CBEF samples were higher for sublegal fish (0.29-0.91) than legal fish (0.17-0.89) during every year except 2011. In general, Pf0 was highest in 2006-2007 and 2011-2012. Confidence intervals of Pf0 overlapped the target criterion (0.30) in 2010 for sublegal fish and in 2008 and 2010 for legal fish. Only 2010, with a 58% chance of Pf0 exceeding the target, exhibited a greater than 1% chance of sublegal fish meeting the nutritional target. CBEF-based estimates of Pf0 for legal Striped Bass during October-November 2008 and 2010 had 44% and 100% chances of exceeding the target criterion; remaining years had a 1% or less chance (Table 10).

During October-November, 2006-2012, few sublegal Striped Bass collected by CBEF had body fat scores greater than 1 (Table 10). Sublegal fish with body fat scores above 1 were not detected by CBEF in 2006, 2009, 2011, and 2012, and only one of the remaining three estimates of Pff (Pff in 2010 = 0.05)

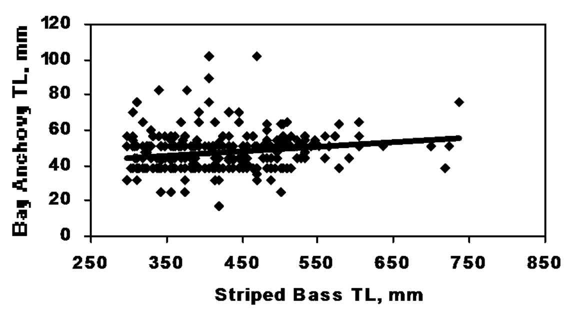

Figure 3. Lengths (TL, mm) of Bay Anchovy consumed and Striped Bass that consumed them during October-November, 2006-2012.

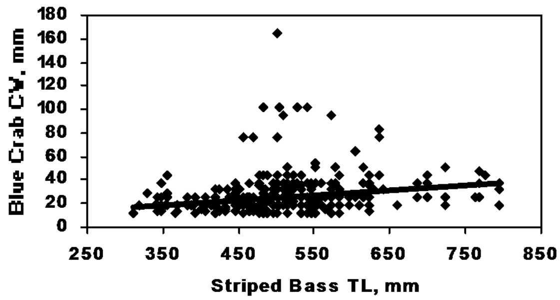

Figure 4. Carapace width of Blue Crab consumed and TL of Striped Bass that consumed them during October-November, 2006-2012.

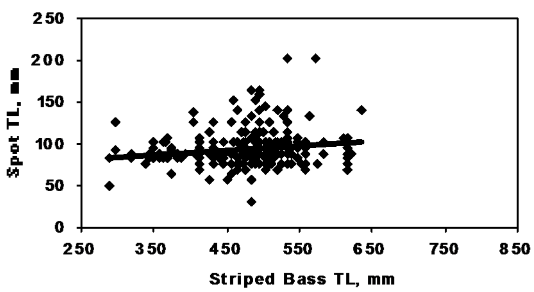

Figure 5. Lengths (TL, mm) of Spot consumed and Striped Bass that consumed them during October-November, 2006-2012

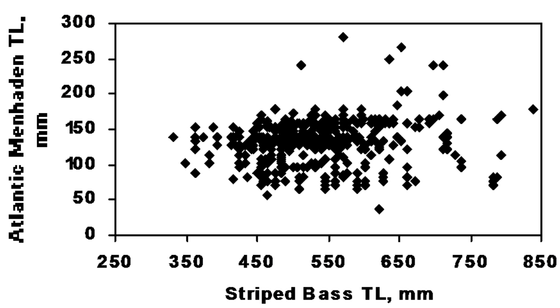

Figure 6. Lengths (TL, mm) of Atlantic Menhaden consumed and Striped Bass that consumed them during October-November, 2006-2012.

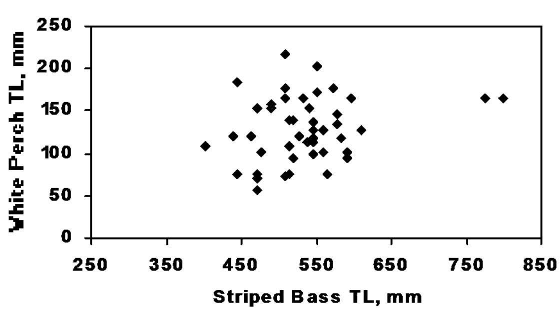

Figure 7. Lengths (TL, mm) of White Perch consumed and Striped Bass that consumed them during October-November, 2006-2012.

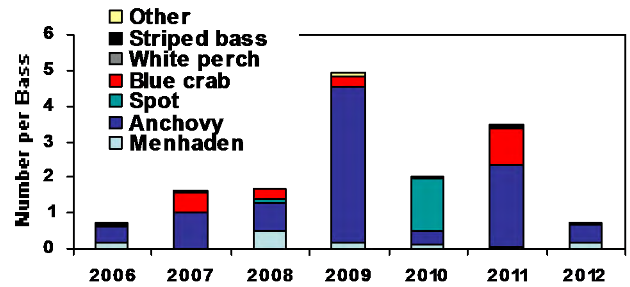

Figure 8. Number of items consumed per sublegal (286-456 mm) Striped Bass during October-November by year.

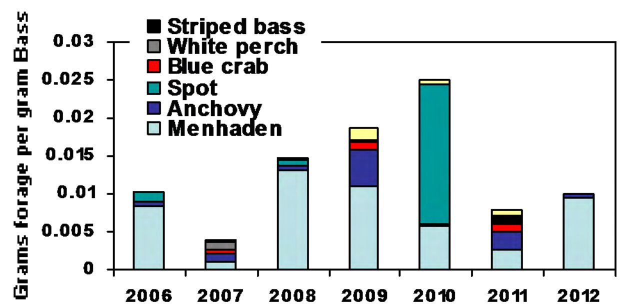

Figure 9. Weight of items consumed per gram of sublegal (286-456 mm) Striped Bass during October-November. Yellow segments of bars indicate "Other" forage.

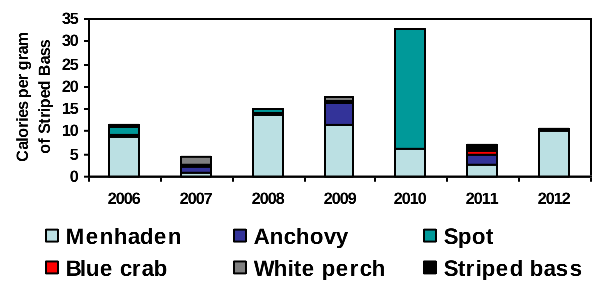

Figure 10. Calories provided for sublegal (286-456 mm) Striped Bass during October-November, by major item.

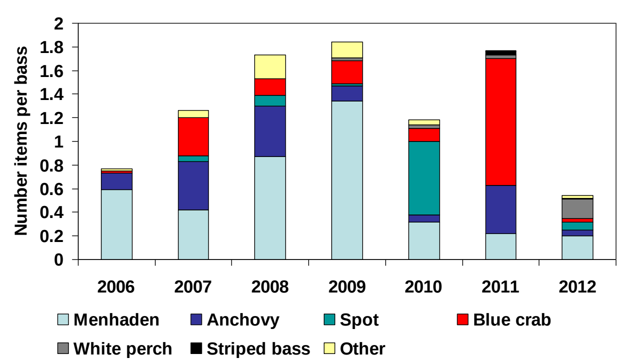

Figure 11. Number of items consumed per legal (457-864 mm) Striped Bass during October-November.

Year	N no fat	N	Pf0	SD	Upper 95%	Lower 95%	P < 30%	Count <1	Pff	SD	Upper 95%	Lower 95%
						CBEF	Sublegal					
2006	96	118	0.81	0.04	0.88	0.74	0%	0	0	0	0	0
2007	69	76	0.91	0.03	0.97	0.85	0%	1	0.013	0.013	0.04	0
2008	14	29	0.48	0.09	0.66	0.3	1%	1	0.034	0.034	0.1	0
2009	51	99	0.52	0.05	0.62	0.42	0%	0	0	0	0	0
2010	32	112	0.29	0.04	0.37	0.21	58%	6	0.054	0.021	0.1	0.01
2011	64	74	0.86	0.04	0.94	0.78	0%	0	0	0	0	0
2012	31	47	0.66	0.07	0.8	0.52	0%	0	0	0	0	0
						CBEF	Legal					
2006	26	49	0.53	0.07	0.67	0.39	0%	6	0.122	0.05	0.21	0.03
2007	120	203	0.59	0.03	0.66	0.52	0%	26	0.128	0.02	0.17	0.08
2008	62	205	0.3	0.03	0.36	0.24	44%	28	0.137	0.02	0.18	0.09
2009	89	240	0.37	0.03	0.43	0.31	1%	35	0.146	0.02	0.19	0.1
2010	51	302	0.17	0.02	0.21	0.13	100%	32	0.106	0.02	0.14	0.07
2011	287	323	0.89	0.02	0.92	0.86	0%	0	0	0	0	0
2012	159	300	0.53	0.03	0.59	0.47	0%	10	0.033	0.01	0.05	0.01
						FWHP	Sublegal					
2006	193	275	0.7	0.03	0.76	0.65	0%	13	0.047	0.013	0.07	0.02
2007	32	36	0.89	0.05	0.99	0.79	0%	0	0	0	0	0
2008	65	213	0.3	0.03	0.37	0.24	40%	48	0.225	0.029	0.28	0
2009	141	279	0.5	0.03	0.56	0.45	0%	44	0.158	0.022	0.2	0.11
2010	54	138	0.39	0.04	0.47	0.31	1%	31	0.225	0.036	0.29	0.16
2011	227	287	0.79	0.02	0.84	0.74	0%	19	0.066	0.015	0.09	0.04
2012	185	224	0.83	0.03	0.88	0.78	0%	11	0.049	0.014	0.08	0.02
						FWHP	Legal					
2006	135	241	0.56	0.03	0.62	0.5	0%	0	0	0	0	0
2007	79	140	0.56	0.04	0.65	0.48	0%	5	0.036	0.016	0.07	0
2008	1	118	0.01	0.01	0.03	-0.01	100%	44	0.373	0.045	0.46	0.29
2009	58	218	0.27	0.03	0.32	0.21	85%	18	0.083	0.019	0.12	0.05
2010	54	215	0.25	0.03	0.31	0.19	93%	94	0.437	0.034	0.5	0.37
2011	144	204	0.71	0.03	0.77	0.64	0%	67	0.328	0.033	0.39	0.26
2012	119	212	0.56	0.03	0.63	0.49	0%	73	0.344	0.033	0.41	0.28

Table 10. Summary of Striped Bass CBEF (Chesapeake Bay Ecological Foundation) and FWHP (MD DNR Fish and Wildlife Health Program) body fat indicators, by size category. N no fat = number without body fat; N = number examined; Pf0 = proportion without body fat; SD = standard deviation of the proportion; Upper 95% = upper 95% confidence interval; Lower 95% = lower 95% confidence interval; P < 30% = chance of being above the target body fat criterion ($\geq$ 30% of sample without body fat); Count < 1 = count of body fat indices greater than 1; Pff = proportion of fish with body fat indices >1.

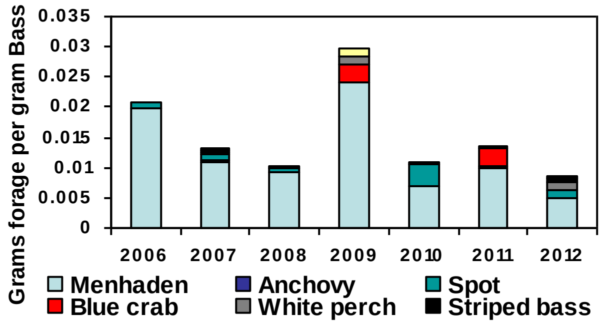

Figure 12. Weight of items consumed per gram of legal (457-864 mm) Striped Bass during October-November. Yellow segments of bars indicate "Other" forage.

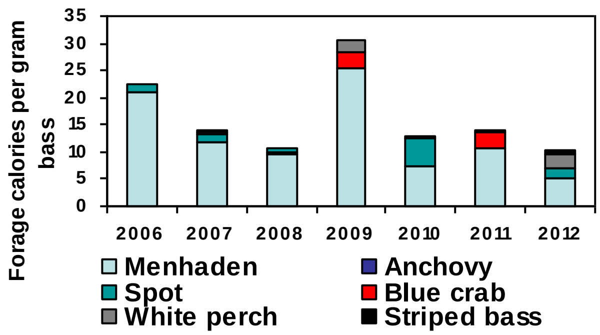

Figure 13. Calories provided for legal (457-864 mm) Striped Bass during October-November, by major item.

was significantly different from zero based on 95% CI overlap. Body fat scores greater than 1 were detected by CBEF for legal Striped Bass during every October-November but 2011. Estimates of Pff for legal fish were significantly different from zero for every year except 2011 (Pff = 0) based on 95% CI overlap. CBEF -based estimates of Pff for legal fish during October-November were similar (0.11-0.15) during 2006-2010 and were lower in 2011-2012 (0 and 0.03; Table 10).

Some general patterns in FWHP-based estimates of Pf0 in mid-Bay during fall were similar to CBEF-based estimates for both size classes of Striped Bass (Table 10). Estimates of Pf0 from FWHP samples were higher for sublegal fish (0.30-0.89) than legal fish (0.01-0.71) during every year. In general, Pf0 was highest in 2006-2007 and 2011-2012 in both sets of samples. However, there were differences in which years met the target criterion. Confidence intervals of FWHP-based Pf0 for sublegal fish overlapped the target criterion (0.30) in 2008, while CBEF-based estimates only overlapped in 2010. Confidence intervals (95%) of FWHP-based Pf0 overlapped the target criterion in 2008-2010 for legal fish (2008 and 2010 for CBEF-based estimates) and there was a high chance of meeting or exceeding the target criterion (85-100%) in those years (Table 10).

Sublegal Striped Bass with body fat scores greater than 1 occurred more frequently in FWHP collections from mid-Bay during October-November, 2006-2012, than CBEF collections (Table 10). FWHP-based estimates of Pff for sublegal fish were different from 0 based on 95% CI overlap for all years except 2007, in contrast to four years of CBEF-based estimates. Estimates of Pff of sublegal fish based on FWHP sampling were low (0.0–0.07) during 2006-2007 and 2011, and higher (0.16-0.23) during the remaining years. Estimates of Pff for legal Striped Bass in FWHEP collections appeared split between two levels during October-November: 0.0-0.08 (2006-2007 and 2009) and (0.33-0.44); CBEF-based estimates of Pff were never higher than 0.15 (Table 10).

Estimates of Pf0 of sublegal and legal Striped Bass were positively and significantly correlated within both data sets (CBEF, r = 0.79, α = 0.03 and FWHP, r = 0.93, α = 0.002). Estimates of Pff were significantly correlated among size classes for FWHP estimates (r = 0.79, α = 0.03), but were not for CBEF estimates (r = 0.32, α = 0.48).

CBEF-based estimates of Pf0 were significantly related to FWHP estimates (r^2 = 0.76, α = 0.0004, df = 12; Figure 14). The relationship was described by the equation:

$$Pf0_F = (0.94 \cdot Pf0_C) - 0.0047;$$

where $Pf0_F$ = proportion of fish without body fat estimated by FWHP and $Pf0_C$ = proportion of fish without body fat estimated by CBEF. Standard errors of the slope and intercept were 0.15 and 0.09, respectively. The slope of this relationship was not significantly different from 1 (t-test, α = 0.67), and the intercept was not significantly different from 0 (α = 0.96). These results supported the hypothesis that there was a 1:1 relationship between CBEF and FHWP estimates of Pf0.

CBEF-based estimates of Pff were significantly related to FWHP estimates (r^2 = 0.44, α = 0.01, df = 12; Figure 15). The relationship was described by the equation:

$$Pff_F = (2.24 \cdot Pff_C) - 0.07;$$

where Pff_F = proportion of fish with body fat scores > 1 estimated by FWHP and Pff_C = proportion of fish with body fat scores > 1 estimated by CBEF. Standard errors of the slope and intercept were 0.73 and 0.06, respectively. The slope of this relationship was not significantly different from 1 (t-test, α = 0.12), and the intercept was not significantly different from 0 (α = 0.23). While these results supported the hypothesis that there was a 1:1 relationship between CBEF and FHWP estimates of Pf0, the regression slope was imprecisely estimated (95% CI = 0.65-3.84). Examination of residuals indicated a potential outlier and highly influential point (FWHP Pff of legal fish = 0.80 at CBEF Pff = 0.13). A second regression was run with this point removed. In this analysis, CBEF-based estimates of Pff were still significantly related to FWHP estimates ($r^2 = 0.46$, $\alpha = 0.01$, df = 11; Figure 15). The relationship was described by the equation:

$Pff_F = (1.28 \cdot Pff_C) - 0.09$.

Standard errors of the slope and intercept were, 0.41 and 0.03, respectively. The slope of this relationship was not significantly different from 1 (t-test, $\alpha = 0.50$), although the estimate of the slope was still imprecise (95% CI = 0.39-2.18). However, the intercept was significantly different from 0 ($\alpha = 0.012$), indicating CBEF estimates of Pff were negatively biased. Examination of residuals did not suggest outliers or influential points.

Nutritional status ($Pf0_C$ or $Pf0_F$) of sublegal Striped Bass during October-November, 2006-2012, was directly related to ∑MWi and∑ MCi. Linear regressions of ∑MWi and ∑MCi versus $Pf0_C$ for sublegal fish

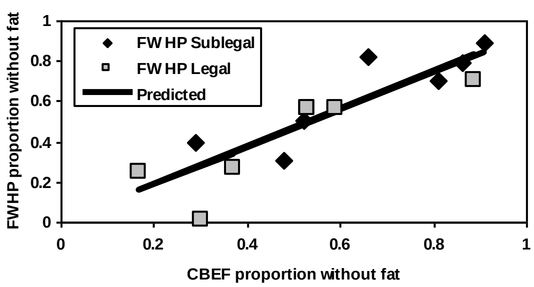

Figure 14. Relationship of CBEF and FWHP estimates of annual proportions of sublegal and legal Striped Bass without visceral body fat during October-November, 2006-2012. Sublegal and legal size groups have been designated by separate symbols, but regression prediction is based on analysis of both sets of data together.

were negative and significant ($r^2 = 0.91$, P = 0.0009 for ∑MWi, Figure 16 and $r^2 = 0.85$, P = 0.003 for ∑MCi, not shown). FWHP estimates did not quite fit as well, but were still significant ($r^2 = 0.74$, P = 0.01 for ∑MWi, Figure 16 and $r^2 = 0.58$, P = 0.05 for ∑MCi, not shown).

Significant relationships were not detected between $Pf0_C$ or $Pf0_F$ and ∑MWi or ∑MCi for legal fish. Regressions with $Pf0_C$ for legal fish did not suggest a relationship ($r^2 = 0.0006$, P = 0.96 for ∑MWi, Figure 17 and $r^2 = 0.006$, P = 0.87 for ∑MCi, not shown), nor did regressions with $Pf0_F$ ($r^2 = 0.09$, P = 0.51 for ∑MWi, Figure 17 and $r^2 = 0.001$, P = 0.94 for ∑MCi, not shown).

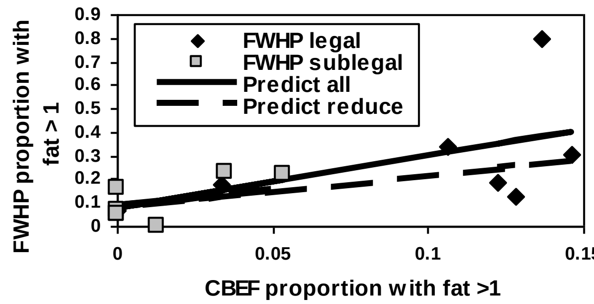

Figure 15. Relationship of CBEF and FWHP estimates of annual proportions of sublegal and legal Striped Bass with visceral body fat scores > 1 during October-November, 2006-2012. Sublegal and legal size groups have been designated by separate symbols, but regression prediction is based on analysis of both sets of data together. Predict all is the predicted line from the regression with all points, including an outlier (red diamond). Predict reduce is the predicted line with the outlier removed.

Estimates of $Pf0_C$ for legal fish in Choptank River and the mainstem Bay were not different based on 95% CI overlap during five of seven years, but were during 2009 (mainstem Bay $Pf0_C = 0.07$ and Choptank River $Pf0_C = 0.51$) and 2012 (mainstem Bay $Pf0_C = 0.83$ and Choptank River $Pf0_C = 0.20$; Table 11). Estimates of $Pf0_C$ were not different from zero in the mainstem Bay during 2009-2010, but were for all other combinations. Bivariate plots of $Pf0_C$ against ∑MWi for legal fish in each region separately (Figure 18) did not suggest results different from the regression with both areas pooled (described above). These estimates of ∑MWi suggested greater foraging success in Choptank River than mainstem Bay since grams of prey consumed per gram of Striped Bass did not exceed 0.02 in mainstem Bay but did reach or exceed this level during three of seven years in Choptank River.

Relative abundances of major pelagic prey (Atlantic Menhaden and Bay Anchovy) were low throughout 2006-2012 (Figure 19), yet these two pelagic prey accounted for 70.5% of sublegal fish diet weight and 82.7% of legal fish diet weight. Atlantic Menhaden had been at this low level during the 1960s before increasing dramatically in the early 1970s; a decline occurred over the following two decades until a nadir was reached in the early1990s that has continued through 2012. Bay Anchovy relative abundance was higher (except for a scattering of poor years) prior to the mid-1990s (Figure 19). Strong year-classes of ma-

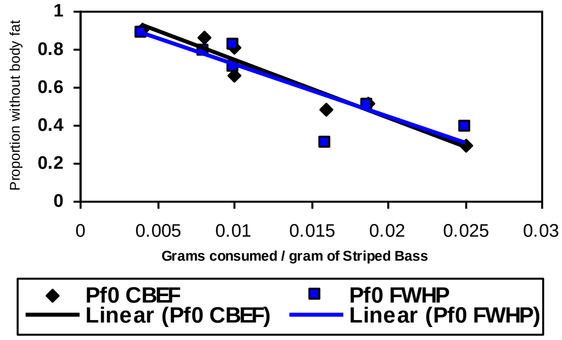

Figure 16. Relationships of grams of prey consumed per gram of sublegal-sized striped bass (286 -456 mm, TL) and proportion of sublegal-sized striped bass without body fat (Pf0) estimated during October-November by CBEF and during fall by FWHP. Lines indicate predicted relationships.

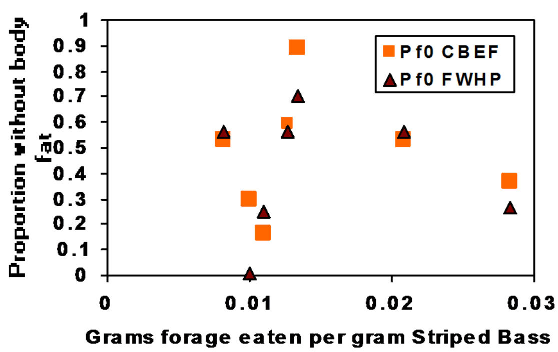

Figure 17. Grams of forage consumed per gram legal-sized striped bass (457-864 mm, TL) versus proportion of legal-sized striped bass without body fat estimated during October-November by CBEF and during fall by FWHP.

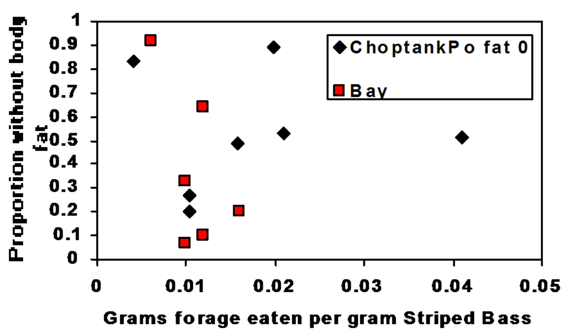

Figure 18. Grams of forage consumed per gram legal-sized striped bass (457-864 mm, TL) versus CBEF-based estimates of proportion of legal-sized striped bass without body fat during October-November in Choptank River and mid-Bay, separately.

jor benthic prey (Spot, Blue Crab, and White Perch) have been interspersed with poor ones during 2006-2012 (Figure 20).

Correlations of sublegal Striped Bass MNi or MWi during October-November with respective forage item relative abundances were positive and significant ($\alpha \leq 0.05$) for Spot (MNi r = 0.96 and MWi r = 0.95) and Blue Crab (MNi, r = 0.79 and MWi, r = 0.80), but not for Atlantic Menhaden or Bay Anchovy (Table 12). Estimates of legal Striped Bass MNi or MWi during October-November were positively and significantly correlated of with relative abundances of Spot (MNi, r = 0.93 and MWi, r = 0.99) and Blue Crab (MNi, r = 0.88 and MWi, r = 0.90). Relative abundances of Atlantic Menhaden, Bay Anchovy, or White Perch were not significantly correlated with feeding success of legal Striped Bass. Significant correlations of feeding success and relative abundance of Spot and Blue Crab may have reflected greater ranges of abundance during 2006-2012 for these species (100% and 41% of respective time-series maximums) than exhibited by Atlantic Menhaden and Bay Anchovy (5% and 20%; Table 12). White Perch had a wide range in relative abundance during 2006-2012 (64% of the time-series maximum) but were not significantly correlated with feeding success of legal Striped Bass. White Perch occurred infrequently in sublegal Striped Bass diets and were not included as major forage for this size class in this analysis.

Discussion

During October-November, 2006-2012, most sublegal and legal Striped Bass in Maryland's mid-Bay region would have been considered vulnerable to starvation based on absence of body fat. Chances of reaching the body fat target (Pf0 $\geq$ 30%) were less than 1% for legal fish in four or five of seven years (FWHP or CBEF estimates, respectively) and six of seven years for sublegal fish (two different years for FWHP and CBEF). In the remaining years, there was a 44-100% chance of legal fish meeting the PF0 target and a 40-58% chance for sublegal fish. Higher body fat scores (body fat index > 1) were not uncommon in legal fish, indicating some were well fed, and were rare in sublegal fish. Nutritional state of sublegal fish was closely related to grams of prey consumed per gram of Striped Bass, but nutritional state of legal fish was not. Occurrence of relatively high Pff for legal fish in some years indicated that prior feeding had been important. Although young-of-year Bay Anchovy, Blue Crab, and Spot, and ages 0 and 1+ White Perch were major prey items during October-November, both grams and calories of prey eaten by both size classes of Striped Bass were usually dominated by age 0 Atlantic Menhaden.

Year	Region	N no fat	N	Pf0	SD	Upper 95%	Lower 95%	Gm/gm
2006	Choptank	26	49	0.53	0.07	0.67	0.39	0.021
2007	Bay	89	140	0.64	0.05	0.74	0.54	0.012
2007	Choptank	31	63	0.49	0.09	0.67	0.32	0.016
2008	Bay	36	109	0.33	0.08	0.48	0.18	0.01
2008	Choptank	26	96	0.27	0.09	0.44	0.1	0.01
2009	Bay	5	75	0.07	0.11	0.29	-0.15	0.01
2009	Choptank	84	165	0.51	0.05	0.62	0.4	0.041
2010	Bay	9	89	0.1	0.1	0.3	-0.1	0.012
2010	Choptank	43	213	0.2	0.06	0.32	0.08	0.01
2011	Bay	158	172	0.92	0.02	0.96	0.88	0.006
2011	Choptank	136	153	0.89	0.03	0.94	0.84	0.02
2012	Bay	159	191	0.83	0.03	0.89	0.77	0.004
2012	Choptank	22	110	0.2	0.09	0.37	0.03	0.016

Table 11. Summary of regional estimates of proportion of legal Striped Bass without body fat. Bay is the mid-portion of mainstem Chesapeake Bay in Maryland. N no fat = number without body fat; N = number examined; Pf0 = proportion without body fat; SD = standard deviation of the proportion; Upper 95% = upper 95% confidence interval; and Gm/gm is the grams of prey consumed per gram of Striped Bass.

		Legal		Sublegal		
GM		MWi	MNi	MWi	MNi	% GM range
Atlantic Menhaden	r	-0.13	-0.58	0.14	0.2	5%
	α	0.781	0.173	0.763	0.664	
Bay Anchovy	r	-0.22	-0.15	-0.61	-0.6	20%
	α	0.632	0.748	0.145	0.158	
Blue Crab	r	0.9	0.88	0.79	0.8	99%
	α	0.0063	0.0089	0.034	0.032	
Spot	r	0.99	0.93	0.96	0.95	41%
	α	<0.0001	0.0026	0.0006	9E-04	
White Perch	r	0.06	-0.18			64%
	α	0.906	0.702			

Table 12. Summary of correlations of MWi (grams of item i consumed per gram Striped Bass) or MNi (numbers of item i consumed per Striped Bass) with Maryland geometric mean (GM) indices of relative abundance for major prey of legal and sublegal fish during October-November, 2006-2012.Parameter r = correlation coefficient and α = level of significance. % GM range = maximum GM during 2006-2012 as a percentage of the full time-series maximum.

Prey indices from Maryland's portion of Chesapeake Bay indicated relative abundances of mid-Bay pelagic prey (Atlantic Menhaden and Bay Anchovy) have been low since the early 1990s. Even though their relative abundance was low, Atlantic Menhaden were dominant prey by weight and calories for both size classes during October-November, 2006-2012. Bay Anchovy often dominated by number in diets of sublegal Striped Bass. Dominance of pelagic prey in Striped Bass diets suggests either larger variations in pelagic prey abundance existed than were measured or availability varied considerably.

In Chesapeake Bay, larger and more abundant Striped Bass resulted from reduced fishing mortality and higher size limits that underpinned management during and since recovery (Richards and Rago 1999). This more conservative management regime, plus strong year-classes of Striped Bass during 1993-2011 (Durell and Weedon 2013), have exacerbated pelagic prey demand manifold (J. Uphoff, unpublished analyses). This demand has not been consistently met since the body fat target during October-November has been attained infrequently.

Reconstructed fall diets of Striped Bass that were smaller or within our sublegal size class during the late 1950s (Griffin 2001; Griffin and Margraf 2003) indicated that Bay Anchovies consumed in Chesapeake Bay averaged 1.4 grams, while those consumed by sublegal Striped Bass during October-November, 2006-2012 averaged 0.5-0.7 grams annually (J. Uphoff, unpublished analysis). Nearly all Bay Anchovies in our diet samples were young-of-year, whereas, mean weight during the 1950s suggests that ages 1+ may

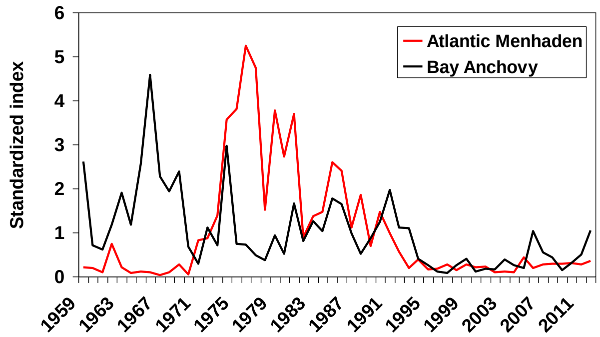

Figure 19. Trends in major pelagic prey of Striped Bass in Maryland's juvenile seine survey. Indices were standardized to their 1989-2012 means.

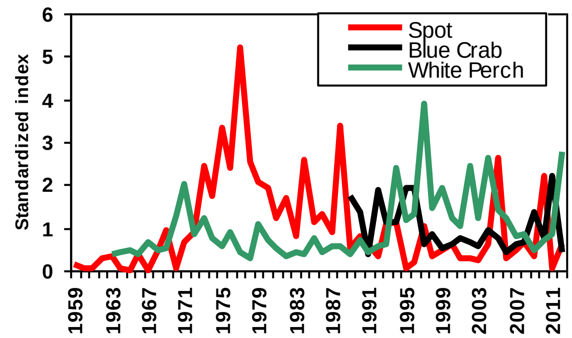

Figure 20. Trends in major benthic prey of Striped Bass in Maryland surveys. Indices were standardized to their 1989-2012 means.

have been more common.

Overton et al. (2009) found that by 1998-2001 Bay Anchovy had become more important to Striped Bass larger than 500 mm TL than in past studies conducted in Maryland's portion of Chesapeake Bay (i.e., Hartman and Brandt 1995b; Griffin and Margraf 2003). Numbers of Bay Anchovies consumed per legal-size Striped Bass during October-November, 2006-2012, ranged from 3% to 54% of that consumed per sublegal fish and tended to be larger than those consumed by sublegal Striped Bass (particularly for Striped Bass > 500 mm; Figure 4-3). Low size limits and higher fishing mortality rates that prevailed into the early 1990s kept Striped Bass that are now legal-size at low levels (Uphoff 2003) and limited competition between size classes. Larger fish tend to forage more efficiently and outcompete smaller fish through greater visual acuity, swimming speed, and experience with the competitive arena (Ward et al. 2006).

In addition to increased intraspecific competition from changes in fisheries management, a long-term decline in the Maryland Bay Anchovy index since 1993 may be linked to declining abundance of the common calanoid copepod *Acartia tonsa* in Maryland's portion of Chesapeake Bay (Kimmel et al. 2012). This drop in *Acartia* was, in turn, linked to rising long-term water temperatures and eutrophication (Kimmel et al. 2012). This decline of *Acartia* coincided with the start of a low abundance regime of age 0 Atlantic Menhaden in Maryland's portion of Chesapeake Bay. Zooplankton is an important component of pre-recruit Atlantic Menhaden diets although direct relationships between zooplankton abundance and Atlantic Menhaden recruitment in Chesapeake Bay have not been detected (Annis et al. 2009).

During 2010, the one year that October-November samples of sublegal Striped Bass collected by CBEF exhibited a reasonable probability of meeting the Pf0 target, a large year-class of Spot provided most of the weight and calories they consumed (75% and 80%, respectively). Successful year-classes of Spot, like the 2010 year-class, were much more frequent during 1973-1988 (8 strong year-classes) than afterwards (2 since 1988). Overton et al. (2009) found that Striped Bass in Chesapeake Bay during 1998-2001 relied more on benthic prey when compared to annual diet studies conducted in the early 1990s (Hartman and Brandt 1995b) and 1950s (Griffin and Margraf 2003). Increasing stable isotope ratios in Striped Bass scales between 1982 and 1998 also indicated increased feeding on benthic prey (Pruell et al. 2003).

The interaction among temperature, fish size, consumption, and metabolism at different levels of activity is complex (Hartman and Brandt 1995b). Strong relationships of Pf0 with ∑MWi and ∑MCi during October-November existed for sublegal fish, while there was little indication of any relationship for legal fish. Estimates of Pff indicated that some legal Striped Bass had been feeding successfully previously and-or elsewhere than where our samples were drawn. Size class-specific estimates of ∑MWi were within similar ranges (0.007-0.028 grams consumed per gram of Striped Bass), but ∑MWi time-series were not correlated between sublegal and legal size classes ($r = 0.12$, $P = 0.79$). Water temperatures during October-November were between 11.0 and 21.0 ºC (measured at mainstem Bay Chesapeake Bay Program monitoring station CB4.2C), sufficient for full growth potential (Hartman and Brandt 1995a), and should not have been metabolically challenging.

Estimates of Pff provided additional indication of differences in nutritional state between sublegal and legal Striped Bass during October-November. Body fat scores above 1 were not uncommon for legal fish in FWHP collections (Pff = 0.25 during 2006-2012 combined). Multiple temporal patterns of change in CBEF high body fat scores of legal fish during October - November were present in plots of body fat scores by date (not shown): high scores were never present (1 year), high scores increased with date (2 years), high scores were stable (3 years), and high scores declined (1 year). The latter two patterns indicated that significant feeding by legal fish had taken place prior to October, so nutritional state reflected this previous history as well as feeding during October -November. FHWP sampled upper, mid-, and lower Chesapeake Bay sequentially and did not have a wide range of dates in the mid-Bay region.

Few sublegal fish had obtained enough nutrition to exhibit high body fat scores during October-November. In the one year that the CBEF estimate of Pff of sublegal fish was different from zero (2010), high body fat scores occurred in mid-to-late November.

Higher growth rates of sublegal Striped Bass would make it necessary for them to devote more lipids towards growth, while both size classes would have been diverting lipids to reproduction and swimming (Ward et al. 2006). During 2006-2012, annual growth increments exhibited by male Striped Bass on the spawning grounds (likely to reside within Chesapeake Bay throughout the year; Cimino and Johnson 2009) at ages corresponding to our sublegal and legal classes were compatible with differences in relationships of Pf0 of the two size classes. Estimates of mean length-at-age were available for male Striped Bass sampled during Maryland spawning stock surveys during 2006-2012 (Giuliano and Versak 2012; Figure 4-21); B. Versak (MD DNR) provided these mean length-at-age estimates. Striped Bass corresponding to our sublegal class were likely to be 2-4 years old since April-May spawning survey mean lengths at age 4 (426-451 mm) fell below 457 mm TL; males 3 years old and older were mature (Cimino and Johnson 2009), so all should have been available for spawning ground samples.

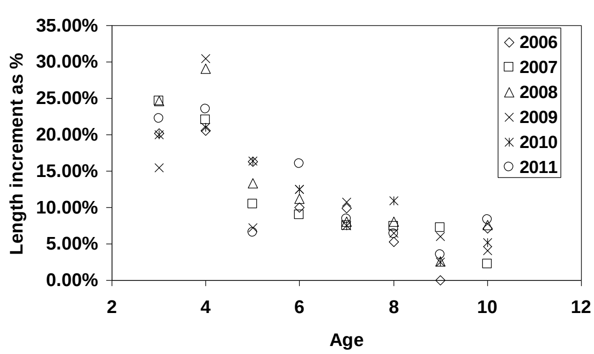

Figure 21. Percent increments of growth in length of male Striped Bass at ages 3-10 from Maryland spawning ground samples (B. Versak, MD DNR, personal communication).

Age 2 was excluded due to low sample sizes and partial maturity that prevented some from being available for spawning ground sampling. We estimated changes in mean length between ages as percent gain to summarize relative growth. Growth increments for ages 3-4 (15.5-30.5%; corresponding to sublegal fish growth from ages 3 to 4 and 4 to 5) were higher than ages 5-10 (0-16.4%; corresponding to legal fish growth; Figure 4-21).

All Striped Bass collected during October-November, 2006-2012, were caught by hook-and-line by volunteers at CBEF expense. This represented an extremely cost-effective diet monitoring approach for MD DNR that could be linked to FWHP monitoring of Striped Bass health. Estimates of Pf0 in mid-Bay by CBEF corresponded well to those made by the FWHP, but there appeared to be negative bias in CBEF estimates of Pff. None-the-less, CBEF estimates did indicate that in some years legal Striped Bass were obtaining sufficient nutrition prior to October-November, while other legal and most sublegal fish were vulnerable to starvation.

Hook-and-line samples are common in marine and estuarine Striped Bass diet studies (Hartman and Brandt 1995c; Nelson et al. 2003; Walter and Austin 2003; Rudershausen et al. 2005; Overton et al. 2008; Overton et al. 2009). Prey availability, schooling behavior of fish, limited search areas and times for fish and fishermen, and non-random behavior of fishermen affect sportfish catchability (Petermen and Steer 1981; Johnson and Carpenter 1994) that could, in turn have affected representativeness of our diet samples. Creel limits (charter boat samples) and collector's permit limits (fishing samples) may have acted to spread samples among locations. On days when catches were difficult to come by, multiple locations would have been fished. If locations where fish could be readily caught were established, samples (catches) from those were capped by these limits. Collections of comparable intensity by alternative gears do not exist in mid-Bay for comparison.

Sampling intensity in mid-Bay during October-November, 2006-2012, was comparable to that employed by Overton et al. (2009) in their regions. In this study, 5-14 separate locations were identified in mid-Bay sampling each year. Overton et al. (2009) sampled 10 to 12 sites in three Chesapeake Bay regions (Maryland and Virginia) during April, 1998 to December, 2001. Eight sites were within the same area we sampled. Over the course of four fall seasons, Overton et al. (2009) sampled 702 Striped Bass comparable to our sublegal and legal classes in their mid-Bay region for an average of about 175 fish. Our study has averaged 311 Striped Bass (both size classes) during October-November, 2006-2012.

Regurgitation of gut contents by hook-and-line caught sampling was possible, although regurgitated contents of Striped Bass stomachs were generally described as slurry (Overton et al. 2008) which may match our "Unknown residue" category. This category was included in estimates of fish without food. "Unknown residue" and "Regurgitated empty" were encountered 6 and 11 times, respectively, for both size classes when all years were pooled. Hook-and-line sampling was not listed by Chipps and Garvey (2007) as a technique that would result in high rates of regurgitation. Sixteen to 57% of sublegal and 37-53% of legal Striped Bass stomachs collect from mid-Bay were empty during October-November, 2006-2012. These estimates were within the range of other Striped Bass diet studies (Walter et al. 2003; Overton et al. 2008; Rudershausen et al. 2005; Overton et al. 2009).

Pope et al. (2001) recommended using caloric values to assess diets, but diet weight may be adequate for evaluating contribution of prey to predator nutrition (Bowen 1989). During October-November, 2006-2012, interpretation of the importance of diet items to Striped Bass nutrition was not very different based on calories (using prey calorie values in Hartman and Brandt 1995a) or weight of food consumed. However, different caloric content could be assigned to the same items. For example, estimates of energy per gram of Spot or White Perch, estimated by Hartman and Brandt (1995a), were 1.3- to 1.7-times, respectively, that of Atlantic Menhaden while estimates for the same items used by Glass and Watts (2009) were 0.5- to 0.6-times the energy content of Atlantic Menhaden (respectively).

Estimates of MWi and ∑MWi provided an index of mean stomach fullness. Expressing diet weight in units of fish body weight has been used to measure stomach fullness among fish of different sizes (Hyslop 1980). Mean stomach fullness (Pope et al. 2001; Chipps and Garvey 2007) has been estimated as the volume of a food item ingested divided by stomach volume. Volume of food can be considered equivalent to weight since specific densities of aquatic organisms are very close to 1.0 (Pope et al. 2001). Stomach volume has been infrequently measured, but Knight and Margraf (1982) and Pope et al. (2001) found that stomach volume could be related to fish length for Walleye *Sander vitreus* and Largemouth Bass *Micropterus salmoides* using allometric equations analogous to those for length-weight:

$$S = a \cdot L^b;$$

where S = stomach volume, L = fish length, a is a scaling coefficient and b is a shape parameter. Pope et al. (2001) believed that assessment of mean stomach fullness based on fish weight would provide similar results to an assessment based on length. Unfortunately, neither study compared the relationship of stomach volume to fish weight, but it may not be unreasonable to assume that stomach volume is linearly related to fish weight (or nearly so) since the same form of functional relationship applies well to length. Parameter b of the length – stomach volume equation estimated by Pope et al. (2001) for Largemouth Bass (3.248) was within the range estimated for length-weight for Largemouth Bass studies (2.73-3.48) cited in Fishbase (Froese and Pauly 2013), indicating the possibility of similar rates of increase with length. The estimate of b made for Walleye stomach volume by Knight and Margraf (1982), 2.56, was below the range of estimated rates of increase of weight (3.08-3.23) with length cited in Fishbase (Froese and Pauly 2013). Neither Knight and Margraf (2001) nor Pope et al. (2001) provided standard errors for parameter b to construct confidence intervals.

We calculated our diet metrics from our entire October-November size class sample for each year and these estimates do not have corresponding variance estimates (Chipps and Garvey 2007). Feeding indices (MNi, MWi, and MCi) summarized diet at population scale without a measure of individual variation. The ultimate objective of this job is to understand the dynamics of consumption and their effect on nutritional state at different times of year and these estimators have provided a good starting point. In the future, we anticipate estimating similar metrics from means based on individual fish that will provide variance estimates. We will expand the seasons in the analysis until we have year-round coverage of diet and nutrition dynamics.

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